Bonneville Power Administration, USACE – Portland District, USACE – Walla Walla District, and Grant County Public Utility District

Research, Monitoring, and Evaluation of Avian Predation on Salmonid Smolts in the Lower and Mid-Columbia River

2013 Final Annual Report

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This 2013 Final Annual Report has been prepared for the Bonneville Power Administration, the U.S. Army Corps of Engineers, and the Grant County Public Utility District for the purpose of assessing project accomplishments. This report is not for citation without permission of the authors.

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TABLE OF CONTENTS

EXECUTIVE SUMMARY………………………………………………………………………… 7

INTRODUCTION…………………………………………………………………………………… 13

STUDY AREA……………………………………………………………………………………… 16

SECTION 1: CASPIAN TERNS…………………………………………………………………… 16

1.1. Preparation and Modification of Nesting Habitat……………………………………… 16
1.1.1. Columbia River Estuary……………………………………………………………… 17
1.1.2. Interior Oregon and Northeastern California……………………………………… 18

1.2. Colony Size and Productivity……………………………………………………………… 19
1.2.1. Columbia River Estuary……………………………………………………………… 19
1.2.2. Columbia Plateau……………………………………………………………………… 21
1.2.3. Coastal Washington…………………………………………………………………… 23
1.2.4. Interior Oregon and Northeastern California……………………………………… 28

1.3. Diet Composition and Salmonid Smolt Consumption…………………………………… 32
1.3.1. Columbia River Estuary……………………………………………………………… 32
1.3.2. Columbia Plateau……………………………………………………………………… 34
1.3.3. Coastal Washington…………………………………………………………………… 36
1.3.4. Interior Oregon and Northeastern California……………………………………… 36

1.4. Predation Rates Based on Salmonid Smolt PIT Tag Recoveries……………………… 38
1.4.1. Columbia River Estuary……………………………………………………………… 42
1.4.2. Columbia Plateau……………………………………………………………………… 44
1.4.3. Coastal Washington…………………………………………………………………… 50
1.4.4. Interior Oregon and Northeastern California……………………………………… 50

1.5. Color Banding and Band Re-sightings…………………………………………………… 51
1.5.1. Columbia River Estuary……………………………………………………………… 51
1.5.2. Columbia Plateau……………………………………………………………………… 53
1.5.3. Interior Oregon and Northeastern California……………………………………… 54

SECTION 2: DOUBLE-CRESTED CORMORANTS…………………………………………… 56

2.1. Nesting Distribution and Colony Size……………………………………………………… 56
2.1.1. Columbia River Estuary……………………………………………………………… 56
2.1.2. Columbia Plateau……………………………………………………………………… 58
2.1.3. Coastal Washington…………………………………………………………………… 61
2.1.4. Interior Oregon and Northeastern California……………………………………… 61

2.2. Nesting Success……………………………………………………………………………………… 62
2.2.1. Columbia River Estuary…………………………………………………………………… 62
2.2.2. Columbia Plateau……………………………………………………………………………. 62
2.2.3. Coastal Washington………………………………………………………………………… 63
2.2.4. Interior Oregon and Northeastern California……………………………………… 63

2.3. Diet Composition and Salmonid Smolt Consumption…………………………………… 63
2.3.1. Columbia River Estuary…………………………………………………………………… 63
2.3.2. Columbia Plateau……………………………………………………………………………. 66
2.3.3. Coastal Washington………………………………………………………………………… 66
2.3.4. Interior Oregon and Northeastern California……………………………………… 66

2.4. Predation Rates Based on Salmonid Smolt PIT Tag Recoveries………………………… 66
2.4.1. Columbia River Estuary…………………………………………………………………… 66
2.4.2. Columbia Plateau……………………………………………………………………………. 68

2.5. Color-banding…………………………………………………………………………………… 68

2.6. Management Feasibility Study………………………………………………………………… 70

SECTION 3: OTHER PISCIVOROUS COLONIAL WATERBIRDS…………………………………… 74

3.1. Distribution……………………………………………………………………………………… 74
3.1.1. Columbia River Estuary…………………………………………………………………… 74
3.1.2. Columbia Plateau……………………………………………………………………………. 76
3.1.3. Coastal Washington………………………………………………………………………… 77
3.1.4. Interior Oregon and Northeastern California……………………………………… 78

3.2. Diet Composition………………………………………………………………………………… 79
3.2.1. Columbia River Estuary…………………………………………………………………… 79
3.2.2. Columbia Plateau……………………………………………………………………………. 80
3.2.3. Coastal Washington………………………………………………………………………… 81
3.2.4. Interior Oregon and Northeastern California……………………………………… 81

3.3. Predation Rates Based on Salmonid Smolt PIT Tag Recoveries………………………… 82
3.3.1. Columbia River Estuary…………………………………………………………………… 82
3.3.2. Columbia Plateau……………………………………………………………………………. 83

ACKNOWLEDGMENTS………………………………………………………………………………… 87

LITERATURE CITED……………………………………………………………………………………… 88

PROJECT PUBLICATIONS AND SUBMITTED MANUSCRIPTS……………………………………… 98

PROGRAM FUNDING…………………………………………………………………………………………… 102

MAPS……… 105

FIGURES……………………………………………………………………………………………………… 111

TABLES…… 184

APPENDIX A: PIT TAG DEPOSITION STUDIES……………………………………………………… 193

APPENDIX B: PREVIOUSLY REPORTED CASPIAN TERN PREDATION RATES
ADJUSTED FOR SMOLT PIT TAG DEPOSITION RATE, 2007-2013…………………………… 208

APPENDIX C: FORAGING BEHAVIOR AND DISPERSAL PATTERNS OF
CASPIAN TERNS NESTING ON GOOSE ISLAND, POTHOLES RESERVOIR……………………. 211

APPENDIX D: SUPPLEMENTAL FIGURES AND TABLES……………………………………………. 239

EXECUTIVE SUMMARY

The primary objectives of this project in 2013 were to (1) evaluate management initiatives implemented to reduce predation on juvenile salmonids (*Oncorhynchus* spp.) by Caspian terns (*Hydroprogne caspia*) nesting on East Sand Island in the Columbia River estuary, including the monitoring of alternative Caspian tern nesting islands built by the Corps outside the Columbia River basin; (2) collect, compile, and analyze data needed to assist in completion of the NEPA analysis required for management of (a) double-crested cormorants (*Phalacrocorax auritus*) nesting on East Sand Island and (b) Caspian terns nesting at colonies in the Columbia Plateau region; (3) investigate the numbers of other piscivorous colonial waterbirds (i.e., Brandt's cormorants *P. penicillatus*, California brown pelicans *Pelecanus occidentalis californicus*, American white pelicans *P. erythrorhynchos*, and gulls *Larus* spp.) that use the Columbia River to nest or roost and assess their potential impacts on smolt survival; and (4) assist resource managers as technical advisors in the development of plans for long-term management of avian predation on juvenile salmonids from the Columbia River basin, as warranted.

The Caspian tern colony on East Sand Island, the largest for the species in the world, consisted of about 7,400 breeding pairs in 2013. This is an increase from the estimate of 6,400 pairs in 2012, and the first increase since the initiation of habitat reduction on East Sand Island in 2008, when the colony numbered about 10,000 breeding pairs. In addition to the increase in colony size, Caspian terns at this colony were more resilient to disturbances by bald eagles and associated gull depredation on tern eggs and chicks compared to during 2010-2012. The Caspian tern colony on East Sand Island produced about 1,480 fledglings in 2013 (average of 0.20 young raised/breeding pair), a significant increase from 2010-2012 (average of 0 - 0.06 young raised/breeding pair), but lower than in other years during 2001-2009. The average proportion of juvenile salmonids in the diet of Caspian terns during the 2013 nesting season was 32%, similar to 2009-2012. The estimated total smolt consumption by Caspian terns nesting at East Sand Island in 2013 was 4.6 million (95% c.i. = 3.9 - 5.3 million), similar to 2012. Recoveries of smolt passive integrated transponder (PIT) tags on the Caspian tern colony at East Sand Island indicated that tern predation rates on salmonid smolts that were interrogated passing Bonneville Dam on the Columbia River or Sullivan Dam on the Willamette River were similar in 2013 and 2012. Also similar to previous years, tern predation rates were significantly higher on steelhead populations (8.6 – 12.5%, depending on the population) compared with salmon populations (0.6 – 1.4%, depending on the population). Despite the increase in the size of the tern colony in 2013 compared to 2012, predation rates on ESA-listed steelhead and salmon populations have trended lower since tern habitat reductions were initiated in 2008.

Caspian tern management actions in the Columbia River estuary continued in 2013. The U.S. Army Corps of Engineers, Portland District (Corps) maintained 1.58 acres of suitable nesting habitat for Caspian terns on East Sand Island, the same area of habitat as in 2012, and 32% of the area of nesting habitat provided during 2001-2007. The restriction in available tern nesting habitat on East Sand Island to 1.58 acres caused Caspian terns to nest at an average density of 1.17 nests/m^2, an increase from 1.06 nests/m^2 in 2012, and the highest tern nesting density so far observed in the Columbia River estuary. In addition, several hundred pairs of Caspian terns attempted to nest in three discrete satellite colonies on the upper beach of East Sand Island, adjacent to the 1.58-acre area of designated tern nesting habitat; however, no young were successfully raised in these satellite colonies. Passive deterrence measures (stakes, ropes, and flagging) installed by the Corps to dissuade Caspian terns from nesting on areas of the upper beach near the main colony along with tidal inundations of some nest sites were effective in limiting the formation and size of satellite colonies.

The Corps has constructed nine islands as alternative Caspian tern nesting colony sites since early 2008, six in interior Oregon and three in the Upper Klamath Basin region of northeastern California. Two of these islands were not available for tern nesting in 2013, and one is no longer being monitored for Caspian tern nesting activity. Of the six islands that were monitored for Caspian tern nesting activity, five supported nesting Caspian terns. A combined total of over 1,100 breeding pairs of Caspian terns nested at these five alternative colony sites in 2013, a 50% increase from 2012. Estimated productivity was low among the five sites, however, ranging from an average of 0 to 0.37 young raised/breeding pair, depending on the site. In 2013, mammalian and avian nest predators, displacement by other colonial waterbird species (i.e., California gulls *L. californicus*, American white pelicans), drought, adverse weather condition, and likely low forage fish availability (due to drought), limited Caspian tern colony formation, colony size, and nesting success at one or more of the alternative colony sites. A substantial number of Caspian terns that were banded at the colony on East Sand Island in the Columbia River estuary, however, did use the alternative colony sites created by the Corps; 57 terns banded in the Columbia River estuary were seen at the alternative colony sites in interior Oregon and 110 were seen at the alternative colony sites in the Upper Klamath Basin during the 2013 nesting season. Based on estimated movement rates (5.3%) calculated from Caspian terns banded as adults, 684 Caspian terns (including both banded and unbanded terns) were estimated to have moved from East Sand Island to alternative colony sites in 2013.

To further reduce the impacts of predation by Caspian terns nesting at East Sand Island on salmonid stocks from the Columbia River basin, more terns will need to be relocated to colonies outside the basin; the management objective is to reduce the size of the East Sand Island tern colony to 3,125 - 4,375 breeding pairs, less than 45% of its pre-management size (ca. 10,000 breeding pairs), while attracting the displaced Caspian terns to alternative colony sites.

The double-crested cormorant colony on East Sand Island consisted of about 14,900 breeding pairs in 2013. This is the largest double-crested cormorant colony ever recorded on East Sand Island, and is about 15% larger than it was during 2011-2012. This one colony likely includes more than 40% of the breeding population of double-crested cormorants in western North America, and is the largest known breeding colony of the species anywhere. In addition to double-crested cormorants, an estimated 1,550 pairs of Brandt's cormorants nested in the cormorant colony on East Sand Island in 2013. Brandt's cormorants first nested in this mixed-species colony in 2006, and numbers increased each year through 2012, when 1,680 breeding pairs were counted.

Juvenile salmonids represented about 11% (by biomass) of the double-crested cormorant diet in 2013, compared to about 20% in 2012. Our estimate of total smolt consumption by double-crested cormorants nesting on East Sand Island in 2013 is 16.3 million smolts (95% c.i. = 11.4 – 21.1 million), not significantly different from the number of smolts consumed by cormorants from this colony in 2012. The majority of these consumed smolts (about 11.4 million or 70%) were sub-yearling Chinook salmon (*O. tshawytscha*) smolts consumed predominantly in the latter portion of the cormorants breeding season (June – August), as was the case in 2012. Estimated consumption of spring migrating smolts (coho [*O. kisutch*], yearling Chinook, and sockeye [*O. nerka*] salmon along with steelhead [*O. mykiss*]), however, was 4.8 million smolts (95% c.i. = 3.8 – 5.8 million), significantly less than consumption of spring migrants in 2012 (8.1 million smolts [95% c.i. = 6.2 – 9.9 million smolts]). This approximately 41% reduction in consumption of spring migrants occurred despite a 21% increase in peak cormorant colony size, suggesting a pronounced decline in proportion of spring migrants in the diet of double-crested cormorants during the 2013 nesting season.

As in other recent years, estimates of total smolt consumption by double-crested cormorants nesting on East Sand Island in 2013 were significantly higher than that of Caspian terns nesting on East Sand Island. During 2004 – 2013, estimates of total annual smolt consumption by the East Sand Island double-crested cormorant colony have varied widely, from a low of 2.4 million smolts to a high of 20.5 million smolts (mean = 12.3 million). This large inter-annual variability in smolt consumption (coefficient of variation {CV} = 49%) has occurred over a period of relatively stable colony size (10,950 – 14,900 breeding pairs; CV = 9%) and has closely tracked the proportion of the cormorant diet that was salmonid smolts (2 – 20% of biomass consumed; CV = 47%); the proportion of smolts in the cormorant diet is an important input parameter in the bioenergetics calculations.

During 2004 - 2013, the type of salmonid consumed in the largest numbers by double-crested cormorants nesting at East Sand Island was sub-yearling Chinook salmon (ca. 7.8 million smolts/year), followed by coho salmon, steelhead, and yearling Chinook salmon (ca. 2.4, 1.1, and 1.0 million smolts/year, respectively). Recoveries of smolt PIT tags on the East Sand Island cormorant colony in 2013 indicated that population- or ESU-specific predation rates ranged from 0.7% to 2.9% for populations originating upstream of Bonneville Dam on the Columbia River or upstream of Sullivan Dam on the Willamette River. Despite the increase in the size of the cormorant colony in 2013, the ESU-specific predation rates measured in 2013 were some of the lowest recorded since 2007.

Similar to consumption estimates (number of fish consumed based on bioenergetics calculations), cormorant predation rates on particular populations or ESUs of salmonids based on smolt PIT tag recoveries have been highly variable among salmonid populations and among years. As demonstrated by the data collected in 2013, variability in the impacts on salmonids from cormorant predation cannot be explained by differences in colony size alone. Factors driving the large inter-annual variation in impacts of cormorant predation (smolt consumption and predation rates) are poorly understood, but may include environmental conditions in the estuary, the abundance and arrival timing of marine forage fish in the estuary, differences in cormorant nesting chronology and success, and/or other biotic and abiotic factors that influence cormorant feeding behavior.

In 2013, the USACE expanded a pilot study initiated in 2011 to test possible strategies for limiting the size of the East Sand Island cormorant colony. Prior to the nesting season, two 8-foot-high privacy fences were built to bisect the colony. These fences visually separated 4.0 acres (25%) of the ca. 16 acres of available cormorant nesting area at the west end of East Sand Island. We used human disturbance to haze cormorants during the nest initiation period, and were successful in dissuaded them from using areas outside the 4.0-acre designated area in 2013. Some double-crested cormorants were satellite-tagged (n = 83) to follow their post-hazing movements to prospective new nesting sites. About 96% of these tagged cormorants (80/83) dispersed from the East Sand Island colony after tagging, and of these about 96% eventually returned to East Sand Island (73/76) and attempted to nest there. Tagged cormorants dispersing from East Sand Island during the nesting season were detected at colonies and roost sites (1) elsewhere in the Columbia River estuary (n = 76), (2) on the lower Columbia River below Bonneville Dam (n = 27), (3) on the outer Washington coast (including Willapa Bay and Grays Harbor; n = 21), and (4) in Puget Sound (n = 1). No double-crested cormorants satellite-tagged on East Sand Island early in the 2013 nesting season were detected along the coast of Oregon during the nesting season.

Native piscivorous colonial waterbirds that nest in the Columbia Plateau region include Caspian terns, double-crested cormorants, American white pelicans, California gulls, and ring-billed gulls (*L. delawarensis*). Of these, Caspian terns have been identified as the single most significant avian predator (per capita) in the Columbia Plateau region on salmonid smolts, particularly ESA-listed steelhead populations from the Upper Columbia River and Snake River. Total numbers of Caspian terns nesting in the Columbia Plateau region declined from ca. 870 breeding pairs during 2005-2012 to ca. 775 breeding pairs in 2013, distributed among five breeding colonies. In 2013, the two largest Caspian tern colonies in the Columbia Plateau region were at Crescent Island (395 breeding pairs) on the mid-Columbia River and at Goose Island (340 breeding pairs) on Potholes Reservoir, WA. The size of both of these Caspian tern colonies declined from 2012 to 2013, but nesting success at both colonies increased in 2013. A small number of Caspian terns (about 26 breeding pairs) established nests at the Blalock Islands in the mid-Columbia River, but nesting success was quite limited. We observed a small number of banded

Caspian terns that were originally banded as adults on East Sand Island in the Columbia River estuary, where management actions have been implemented, at the Goose Island (n = 1) and Crescent Island (n = 4) colonies in 2013. Prior to 2011, when tern management intensified at East Sand Island, movement to the Columbia Plateau region by banded adult Caspian terns that previously nested on East Sand Island had not been documented.

Estimates of Caspian tern predation rates on salmonids based on smolt PIT tag recoveries on tern colonies in the Columbia Plateau region indicated that impacts were greatest on the upper Columbia River steelhead population (14.9% depredated by terns from the Goose Island colony) and on the Snake River steelhead population (2.8% depredated by terns from the Crescent Island colony). Predation rates by Goose Island terns on upper Columbia River yearling Chinook were also notable (2.1%), but significantly lower than predation rates on steelhead. Predation rates by Caspian terns nesting at the small colony in the Blalock Islands were an order of magnitude less than those of terns nesting at Goose and Crescent islands, but steelhead were still highly susceptible to predation by terns from this colony.

A total of 23 adult Caspian terns nesting at Goose Island were marked with GPS tags and tracked during foraging trips over several days. Nearly half of the GPS-tagged terns (n = 11) made foraging trips to the mid-Columbia River, including Wanapum Reservoir, Priest Rapids Reservoir, and Hanford Reach. Of note, four GPS-tagged terns made foraging trips to the lower Snake River, including one tern that exhibited the greatest foraging range ever documented in a breeding Caspian tern: 93 km straight-line distance from the colony.

Management of the Caspian tern colonies at Goose and Crescent islands to reduce their impacts on ESA-listed salmonids is currently under consideration by regional managers. Band re-sighting data indicate high connectivity among Caspian tern colonies in the Columbia Plateau region and colonies elsewhere in western North America from Mexico to Alaska, both inland and along the coast. This suggests that Caspian terns displaced from these two Columbia Plateau colonies may re-nest at existing or newly-created colony sites outside the Columbia River basin.

Total numbers of double-crested cormorants nesting in the Columbia Plateau region decreased slightly in 2013, from about 1,570 breeding pairs during 2012 to about 1,400 breeding pairs at four colonies in 2013; the largest colonies were in the North Potholes Reserve (ca. 800 nesting pairs) and on Foundation Island in the mid-Columbia River (ca. 390 nesting pairs). Numbers of American white pelicans nesting on Badger Island in the mid-Columbia River, a colony that experienced rapid growth during 2004-2011, appear to have stabilized at about 2,100 adults. The numbers of California gulls nesting on Miller Rocks, a colony located just downstream of John Day Dam on the Columbia River, were similar to those observed in recent years (ca. 4,800 adults). Following the abandonment of the large California gull colony on Three Mile Canyon Island (ca. 6,200 adults were counted on-colony in 2009), there was a commensurate increase in the number of California gulls nesting on islands in the Blalock Islands complex during 2012-2013; in 2013 ca. 8,100 nesting gulls (mostly California gulls) were counted on two islands in the Blalock Islands, whereas in 2009 no gulls nested there.

Currently there are no plans to manage colonies of cormorants, gulls, or white pelicans in the Columbia Plateau region based on previous research investigating their relative impacts on survival of juvenile salmonids. Recently-collected smolt PIT tag data, however, casts new light on the impacts of certain gull colonies on smolt survival, in particular the California gull colonies on Miller Rocks and Crescent Island. Deposition rates of smolt PIT tags on nesting colonies (percentage of smolt PIT tags that were consumed by gulls and subsequently deposited by the gull on-colony and used to estimate predation rates) by California gulls were significantly lower than those of Caspian terns and double-crested cormorants. Average on-colony PIT tag deposition rates by California gulls was just 17% (95% c.i. = 14% - 19%) in 2012 and 2013. Incorporation of on-colony deposition rates into predation rate models increased the estimates of colony-wide predation rates on juvenile salmonids by a factor of about 6 for California gulls, compared to previously published estimates. After adjusting for on-colony deposition rates, colony-wide predation rates on juvenile salmonids varied significantly by salmonid ESU and gull colony (range: < 0.1% to 8.5%); the California gull colonies on Miller Rocks and Crescent Island had a much greater impact on smolt survival compared to the California gull or ring-billed gull colonies on Island 20 (on the mid-Columbia River near Richland, WA) and the Blalock Islands. These results suggest that smolt predation rates by gulls nesting at some colonies in the Columbia Plateau region are comparable to, if not higher than, those of Caspian terns and double-crested cormorants nesting at colonies in the Columbia Plateau region.

INTRODUCTION

A Columbia Basin-wide assessment of avian predation on juvenile salmonids (*Oncorhynchus* spp.) indicates that the most significant impacts to smolt survival occur in the Columbia River estuary (BRNW 2005a, 2006a, 2007, 2008, 2009a, 2010a, 2011, 2012, 2013). Caspian terns (*Hydroprogne caspia*) and double-crested cormorants (*Phalacrocorax auritus*) nesting at colonies on East Sand Island in the Columbia River estuary together consumed 6 million to 25 million salmonid smolts annually during 2003 – 2012, based on the sum of the best estimates of total smolt consumption by birds nesting at these two colonies in each year. The magnitude of avian predation in the Columbia River estuary represents about 5-20% of all juvenile salmonids that reach the estuary during out-migration (BRNW 2013a). Estimated smolt losses to piscivorous colonial waterbirds that nest in the Columbia River estuary are more than an order of magnitude greater than those observed elsewhere in the Columbia River basin (BRNW 2013a, Lyons et al. 2011a, Lyons et al. 2011b). Additionally, when compared to the impact of avian predation in the Columbia Plateau region, avian predation in the Columbia River estuary affects juvenile salmonids belonging to every evolutionarily significant unit (ESU) or distinct population segment (DPS; hereafter referred to as ESU) from throughout the Basin that have survived freshwater migration to the ocean, and presumably have a higher probability of returning as adults. For these reasons, management of the colonies of Caspian terns and double-crested cormorants on East Sand Island has the greatest potential to benefit salmonid ESUs from throughout the Columbia River basin that are listed under the U.S. Endangered Species Act (ESA), compared to potential benefits of managing other colonies of piscivorous waterbirds. The Caspian tern colonies on Crescent Island (mid-Columbia River) and Goose Island (Potholes Reservoir), the double-crested cormorant colony on Foundation Island (mid-Columbia River), and the gull (*Larus* spp.) colonies on Miller Rocks and Crescent Island (mid-Columbia River) may be exceptions to this rule; management of these relatively small colonies on or near the mid-Columbia River may benefit certain salmonid ESUs, in particular steelhead (*O. mykiss*; Lyons et al. 2011a, Lyons et al. 2011b, Evans et al. 2012).

Regional fish and wildlife managers called for management action in 1999 to reduce losses of juvenile salmonids to Caspian terns nesting in the Columbia River estuary. A management plan implemented in 2000 sought to relocate the Caspian tern colony on Rice Island, the largest of its kind in the world, to a restored colony site on East Sand Island, 21 km closer to the ocean, where it was hoped terns would consume significantly fewer juvenile salmonids. Over 94% of the nesting Caspian terns shifted from Rice Island to East Sand Island in 2000, where juvenile salmonids comprised 47% of tern prey items, compared to 90% of prey items at Rice Island (Roby et al. 2002). During 2001–2012, all Caspian terns nesting in the Columbia River estuary used East Sand Island, with the exception of three nesting pairs that laid a total of 4 eggs on Rice Island in 2011 (BRNW 2013a). During 2001-2012, estimated consumption of juvenile salmonids by Caspian terns nesting on East Sand Island averaged 5.2 million smolts per year (SD = 0.9 million, n = 12 years), a ca. 58% reduction in annual consumption of salmonid smolts compared to when the Caspian tern colony was on Rice Island (12.4 million smolts in 1998; Roby et al. 2003).

Further management of Caspian terns to reduce losses of juvenile salmonids in the Columbia River estuary is currently in progress; the Records of Decision (RODs) for Caspian tern management in the estuary, signed in November 2006, stipulated the redistribution of approximately 60% of the East Sand Island tern colony to alternative colony sites in Oregon and California (USFWS 2005, 2006). This management is intended to further reduce smolt losses to Caspian terns in the estuary by about 60%, while still maintaining the long-term viability of the Pacific Coast population of Caspian terns. By the end of the 2012 nesting season, the U.S. Army Corps of Engineers – Portland District had constructed nine islands, six in interior Oregon and three in northeastern California, as alternative nesting habitat for Caspian terns nesting on East Sand Island. Concurrent with island construction, the Corps has been gradually reducing the area of suitable nesting habitat for Caspian terns on East Sand Island from 5 acres in 2008 to 1.58 acres in 2012, and hazing terns that attempt to establish new nesting colonies elsewhere in the Columbia River estuary.

The numbers of double-crested cormorants nesting on East Sand Island in the Columbia River estuary have increased dramatically in the last two decades; this growth in colony size appears to have been largely at the expense of other colonies in the region, especially along the coast of Washington and British Columbia (Adkins and Roby 2010). During the period 1997-2012 the cormorant colony on East Sand Island increased 140% to ca. 12,300 breeding pairs, the largest known breeding colony for the species in western North America. Although juvenile salmonids represented ca. 18% of the diet for cormorants (% biomass) and ca. 35% of the diet for terns (% prey items) nesting on East Sand Island in 2010-2012, estimated smolt consumption by cormorants was 4-fold greater than that of Caspian terns during this same time period (i.e., 20 million and 5 million smolts consumed by cormorants and terns, respectively). The large numbers of smolts consumed by the double-crested cormorants nesting at the East Sand Island colony are due to both the larger size of the cormorant colony and the greater food requirements of cormorants relative to Caspian terns. The double-crested cormorant colony on East Sand Island has experienced high nesting success (average of 1.8 young raised/breeding pair per year during 1997-2012), perhaps contributing to increases in colony size and the current level of impact of the cormorant colony on smolt survival.

Resource management agencies have decided that management of the large colony of double-crested cormorants on East Sand Island in order to reduce losses of ESA-listed juvenile salmonids in the Columbia River estuary warrants consideration. Reduction in the size of the double-crested cormorant colony on East Sand Island is one management option under consideration. Feasibility studies to test techniques for dissuading double-crested cormorants from nesting on 6% and 31% of the area used by the East Sand Island colony in 2010 were successful in 2011 and 2012, respectively. Following the success of these pilot studies, resource managers decided to test the feasibility of dissuading cormorants from nesting on a larger proportion of their previous nesting area on East Sand Island.

Breeding colonies of piscivorous colonial waterbirds are not limited to the Columbia River estuary, but are distributed throughout the Columbia River basin. Work to systematically evaluate predation on salmonids by colonial nesting waterbirds in the interior Columbia Basin, or the Columbia Plateau region, began in 1997 (Collis et al. 2002a). The initial focus of this investigation was Caspian tern colonies at Crescent Island (Rkm 510), near the confluence of the Columbia and Snake rivers (Antolos et al. 2005), and in Potholes Reservoir near Othello, WA (Antolos et al. 2004; Maranto et al. 2010). In 2004, comprehensive research was initiated to identify waterbird nesting colonies within the Columbia Plateau region that had the greatest impact on survival of anadromous salmonids from the Columbia and Snake rivers, and to evaluate those impacts over a broad range of environmental conditions and management regimes (Roby 2011, Lyons et al. 2011b). Over 100,000 piscivorous colonial waterbirds, representing five different species nesting at 18 different colonies, were documented nesting in the Columbia Plateau region during 2004-2009 (Roby 2011).

As was the case in the Columbia River estuary, Caspian terns and double-crested cormorants were the two species of piscivorous waterbirds responsible for the majority of losses of salmonid smolts to avian predators in the Columbia Plateau region (Roby 2011; Evans et al. 2012). The Caspian tern colony on Crescent Island (Rkm 510) is one of the largest on the Columbia Plateau at about 420 breeding pairs (BRNW 2013a). Also near the confluence of the Snake and Columbia rivers, Foundation Island (Rkm 519) is home to the largest double-crested cormorant colony on the mid-Columbia River, at nearly 400 breeding pairs (BRNW 2013a). While the size of the Crescent Island Caspian tern colony and Foundation Island cormorant colony have remained fairly stable in recent years, the Goose Island (Potholes Reservoir) Caspian tern colony, located ca. 35 km east of the upper Columbia River, has grown roughly 4-fold from 2004 to 2012, and was the largest colony of Caspian terns in the Columbia Plateau region during 2012 (BRNW 2013a). Salmonid ESU-specific predation rates by birds nesting at these three colonies indicated that steelhead were experiencing higher predation rates compared to other salmonid species. Estimated predation rates on steelhead smolts by Caspian terns nesting at Goose Island/Potholes were as high as 15%, while predation rates on steelhead smolts by Caspian terns nesting at Crescent Island and by cormorants nesting at Foundation Island were as high as 5% and 3%, respectively (Lyons et al. 2011b).
The primary objectives of this project in 2013 were to (1) evaluate management initiatives implemented to reduce predation on juvenile salmonids by Caspian terns nesting on East Sand Island, including the monitoring of alternative Caspian tern nesting islands built by the Corps outside the Columbia Basin; (2) collect, compile, and analyze data needed to assist in completion of the NEPA analysis required for management of (a) double-crested cormorants nesting on East Sand Island and (b) avian predators in the Columbia Plateau region; (3) investigate the numbers of other piscivorous colonial waterbirds (e.g., Brandt's cormorants *Phalacrocorax penicillatus*, California brown

pelicans *Pelecanus occidentalis californicus*, American white pelicans *Pelecanus erythrorhynchos*, and gulls *Larus* spp.) that use the Columbia River to nest or roost and assess their potential impacts on smolt survival; and (4) assist resource managers as technical advisors in the development of plans for long-term management of avian predation on juvenile salmonids from the Columbia River basin, as warranted.

STUDY AREA

The primary focus of our research and monitoring efforts in 2013 were at (1) the Caspian tern and double-crested cormorant colonies on East Sand Island in the Columbia River estuary (Map 1), (2) the Caspian tern and double-crested cormorant colonies in the Columbia Plateau region (Map 1), and (3) six recently constructed islands for nesting Caspian terns in interior Oregon (i.e., Crump Lake in the Warner Valley, East Link impoundment in Summer Lake Wildlife Area, Gold Dike impoundment in Summer Lake Wildlife Area, and Malheur Lake in Malheur National Wildlife Refuge) and northeastern California (i.e., Sheepy Lake in Lower Klamath National Wildlife Refuge, and Tule Lake Sump 1B in Tule Lake National Wildlife Refuge; Map 2).

Additionally, this report provides information on nesting Caspian terns along the Washington Coast; nesting Brandt's cormorants and roosting California brown pelicans on East Sand Island; nesting American white pelicans on Badger Island and the recently-formed colony on Miller Sands Spit in the Columbia River estuary; and various gull colonies in the Columbia River estuary, the Columbia Plateau region, and at Corps-constructed tern islands in interior Oregon and northeastern California; Maps 1 and 2).

SECTION 1: CASPIAN TERNS

1.1. Preparation and Modification of Nesting Habitat

Beginning in 2008, the U.S. Army Corps of Engineers (USACE) implemented management described in the January 2005 Final Environmental Impact Statement (FEIS) and November 2006 Records of Decision (RODs) for *Caspian Tern Management to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* (USFWS 2005, 2006). This management plan, which was developed jointly by the U.S. Fish and Wildlife Service (USFWS; lead), the USACE, and NOAA Fisheries, sought to redistribute the majority of Caspian terns nesting at the colony on East Sand Island in the Columbia River estuary to alternative colony sites (artificial islands) in interior Oregon/California and in the San Francisco Bay area by 2015 (Map 2). The goal of the plan is to reduce Caspian tern predation on out-migrating juvenile salmonids (salmon and steelhead) in the Columbia River estuary, and thereby enhance recovery of salmonid stocks from throughout the Columbia River basin, without negatively affecting the Pacific Coast population of Caspian terns. Thirteen of 20 evolutionarily significant units (ESUs) of Columbia Basin salmonids are currently listed as either threatened or endangered under the U.S. Endangered Species Act (ESA).

The Caspian Tern Management Plan called for the creation of approximately 7-8 acres of new or restored Caspian tern nesting habitat (islands) and to actively attract Caspian terns to nest at these sites. As alternative tern nesting habitat is created or restored, the available nesting habitat for Caspian terns on East Sand Island would be reduced from its initial size (approximately 5 acres) to 1.0-1.5 acres.

The specific objectives of the Plan are to reduce the size of the East Sand Island Caspian tern colony to 3,125-4,375 nesting pairs by limiting the availability of suitable nesting habitat, while providing new nesting habitat for Caspian terns at alternative colony sites outside the Columbia River estuary. These objectives were identified as the preferred alternative in the Final Environmental Impact Statement (EIS) released in early 2005 (USFWS 2005). Terns displaced by habitat reduction on East Sand Island are expected to relocate to alternative colony sites including the eight Corps-constructed tern islands in interior Oregon and northeastern California (i.e., Fern Ridge Reservoir, Crump Lake, Summer Lake Wildlife Area [2 separate islands], Tule Lake NWR, Lower Klamath NWR [2 separate islands], and Malheur NWR). Plans for potentially building additional tern colony sites in the San Francisco Bay area and elsewhere are under consideration.

1.1.1. Columbia River Estuary

As part of the Caspian Tern Management Plan for the Columbia River Estuary, the USACE – Portland District prepared just 1.58 acres of suitable nesting habitat for Caspian terns on East Sand Island in 2013 (Map 3). This 1.58-acre colony area was disked and harrowed to remove encroaching European beach grass and other invasive plants in early March 2013. Unlike in previous years, the colony area was not sprayed with pre-emergent herbicide in the spring of 2013. Without annual restoration of the bare sand nesting habitat that Caspian terns prefer, the East Sand Island tern colony would likely be eliminated within a few years by rapidly encroaching pioneer vegetation. The area of Caspian tern nesting habitat prepared on East Sand Island in 2013 was the same acreage prepared for terns the previous year and a 68% reduction from what had been provided in previous years (5 acres; Map 3).

In early April, before terns initiated nests, vertical fencing with black landscape fabric was erected in rows on bare sand or partially vegetated habitat surrounded the 1.58-acre area prepared for Caspian tern nesting in 2012 and 2013 (Map 3). Rope and flagging were also added between fence rows to enhance the dissuasion outside the 1.58-acre tern colony area on East Sand Island.

As in previous years, Caspian terns began digging nest scrapes near the high tide line on the beaches to the east, north, and south of the large Caspian tern colony at East Sand Island beginning in April. These satellite colonies were outside both the 1.58-acre colony area and the area surrounding the colony area where fencing had been erected to dissuade tern nesting. Throughout the 2013 breeding season, a contractor for the USACE (Newalen LLC) erected additional nest dissuasion materials (i.e., stakes, rope, and flagging) in areas where Caspian terns were observed prospecting for nest sites on the upper beaches surrounding the core colony area on East Sand Island. In total, ca. 2 acres of potential tern nesting habitat was covered with these nest dissuasion materials in 2013. Despite these efforts, some Caspian tern nests were initiated in these satellite colonies, eggs were laid, but no young were successfully raised due to inundation of nests during high tide events.

As part of the Management Plan, Caspian terns that prospect for nest sites on dredged-material disposal islands in the upper Columbia River estuary (i.e., Rice Island, Miller Sands Spit, Pillar Rock Sands) are to be prevented from nesting using active (i.e., human hazing) and passive (i.e., stakes and flagging) dissuasion methods. As was done in previous years, the USACE contracted with Newalen LLC to monitor these sites and conduct active and passive hazing of nesting terns, if needed. Nest dissuasion activities in the upper estuary were not necessary in 2013 because terns were not observed exhibiting pre-nesting behaviors in suitable upland nesting habitat during the breeding season.

1.1.2. Interior Oregon and Northeastern California

By the beginning of the 2013 breeding season, the USACE and its state and federal partners had completed construction of nine islands (a total of 8.3 acres; Table 1) specifically designed for Caspian tern nesting as part of the Caspian Tern Management Plan (USFWS 2005). Two one-acre rock-core islands were built prior to the 2008 breeding season, one at Fern Ridge Reservoir in the Willamette Valley, Oregon, and the other at Crump Lake in the Warner Valley, Oregon. These were followed by the construction of two half-acre islands prior to the 2009 breeding season in Summer Lake Wildlife Area, Oregon (a rock-core island in East Link impoundment and a floating island in Dutchy Lake). Prior to the 2010 breeding season, four additional islands were built: a half-acre rock-core island at Gold Dike impoundment in Summer Lake Wildlife Area, a one-acre silt core island at Orems Unit in Lower Klamath National Wildlife Refuge (NWR), a 0.8-acre floating island at Sheepy Lake in Lower Klamath NWR, and a two-acre rock-core island at Tule Lake Sump 1B in Tule Lake NWR. Most recently, prior to the 2012 breeding season, a one-acre rock-core island was built at Malheur Lake in Malheur NWR. Of these nine tern islands, six were monitored for Caspian tern nesting in 2013; the island at the Orems Unit impoundment was not surrounded by water, the floating island at Dutchy Lake was removed prior to the 2013 breeding season (no longer available for tern nesting), and after four years of no detected Caspian tern nesting attempts at the Fern Ridge Reservoir tern island, monitoring ceased at that site in 2012. Social attraction techniques (i.e., decoys and audio playback systems; Kress 1983, Kress 2000, Kress and Hall 2002, Roby et al. 2002) were used at each of the six monitored tern islands, with the exception of the Crump Lake tern island, to enhance prospects for Caspian terns to nest at each site.

1.2. Colony Size and Productivity

1.2.1. Columbia River Estuary

Methods: The number of Caspian terns breeding on East Sand Island in the Columbia River estuary was estimated using low-altitude, high-resolution aerial photography of the colony taken near the end of the incubation period. The average of 3 direct counts of all adult terns on the colony in aerial photography, corrected using ground counts of the ratio of incubating to non-incubating terns on 12 different plots within the colony area, was used to estimate the number of breeding pairs on the colony at the time of the photography. Confidence intervals for the number of breeding pairs were calculated using a Monte Carlo simulation procedure to incorporate the variance in the multiple counts from the aerial photography and the variance in the ratios of incubating to non-incubating adult terns among the 12 plots. Estimates of breeding pairs were calculated one thousand times using random draws from the sample distributions of numbers of terns on-colony and the ratio of incubating to non-incubating adult terns on plots. Standard error and confidence interval for number of breeding pairs were derived from the resulting distribution.

Fledglings were produced in two waves on East Sand Island in 2013, one in early July and the other in late July. Late nesting terns continued to fledge chicks sporadically into late September, however. Nesting success (average number of young raised per breeding pair) at the East Sand Island tern colony was estimated using aerial photography taken of the colony just prior to these two fledging periods. The average of 3 direct counts of all terns (adults and juveniles) on the colony in aerial photography, corrected using ground counts of the ratio of fledglings to adults on 12 different plots within the colony area, was used to estimate the number of fledglings on the colony at the time of the photography. To estimate nesting success, the total number of fledglings on-colony during these two periods was summed and divided by the number of breeding pairs estimated from the late incubation photo census (see above). Confidence intervals for nesting success were calculated using a Monte Carlo simulation procedure to incorporate the variance in the multiple counts from the aerial photography and the variance in the ratios of fledgling to adults on the plots. Monte Carlo calculations were performed using Visual Basic within Microsoft Excel (Microsoft Corp., Redmond, WA); 1000 iterations were performed and 95% bootstrap percentile limits were used for confidence intervals.

Periodic boat-based and aerial surveys of the dredged material disposal islands in the upper estuary (i.e., Rice Island, Miller Sands Spit, and Pillar Rock Sands) were conducted during the breeding season in order to detect signs of any nesting attempts by Caspian terns.

Results and Discussion: We estimate that 7,387 breeding pairs of Caspian terns (95% c.i. = 6,776 – 7,998 breeding pairs) were nesting on East Sand Island at the peak of nesting activity (mid-June) in 2013 (Figure 1). This total includes 276 breeding pairs that attempted to nest at three satellite colonies located near the high tide line on the beaches to the north, east, and south of the main colony at East Sand Island. The size of the Caspian tern colony on East Sand Island in 2013 was higher than the best estimate of peak colony size in 2012 (6,416 breeding pairs, 95% c.i. = 5,545 – 7,287 breeding pairs; Figure 2). To date, the East Sand Island tern colony continues to be the largest known breeding colony of Caspian terns in the world.

The size of the East Sand Island Caspian tern colony has gradually declined since 2008 until 2013, when colony size increased by nearly 1,000 pairs as compared to the previous year (Figure 2 and Appendix D, Table D1). The overall decline in colony size at East Sand Island tern colony during this period can be attributed to the planned reduction in tern nesting habitat on East Sand Island as part of the Caspian Tern Management Plan for the Columbia River Estuary (USFWS 2005, 2006; see above). During 2008-2012, the amount of nesting habitat prepared for terns on East Sand Island had been incrementally reduced in each year, from approximately 5 acres in 2008 to 1.58 acres in 2012. In 2013, the amount of nesting habitat prepared for terns on East Sand Island remained the same as the previous year (1.58 acres), while nesting density increased to the highest level ever observed for Caspian terns at East Sand Island (1.17 nests/m^2 in 2013; Figure 3 and Appendix D, Table D1). It is likely that suitable nesting habitat for Caspian terns on East Sand Island is limiting, particularly in the last three years. Further reductions in the amount of Caspian tern nesting habitat provided on East Sand Island will be necessary to realize the goal of reducing the size of the East Sand Island tern colony to 3,125 – 4,375 breeding pairs, as prescribed in the Caspian Tern Management Plan.

Using the July 9 aerial photography of the Caspian tern colony, we estimate that ca. 935 fledglings were produced in the first wave of successful nesting at the main colony on East Sand Island. Later counts indicate that an additional ca. 545 fledglings were produced at the main colony from nests that successfully persisted later into the breeding season. No tern chicks were observed at any of the satellite colonies at East Sand Island in 2013 due to nest scrapes with eggs being washed away during high high tide events in mid-June. In total, we estimate that about 1,480 fledglings were produced at the main colony on East Sand Island in 2013. This corresponds to an average nesting success of 0.20 young raised per breeding pair (95% c.i. = 0.15 – 0.25 fledglings/breeding pair). Compared to previous Caspian tern nesting success at East Sand Island and at other Caspian tern colonies in the region, this is considered poor productivity, but productivity at East Sand island in 2013 was quite a bit higher than the average productivity observed at this colony during the previous three years (0.04 fledglings/breeding pair). Nesting success at the East Sand Island Caspian tern colony peaked in 2001 and has trended downward since then (Figure 4). At least two factors have contributed to the decline in productivity of the Caspian tern colony at East Sand Island: (1) ocean conditions and/or high river flows as they influence the availability of marine forage fishes in the estuary and (2) predation on tern nests by gulls, especially during tern colony disturbance events caused by bald eagles (Collar 2013).

Unlike what had been observed in previous years, Caspian terns did not prospect for nest sites at dredged material disposal sites in the upper Columbia River estuary during 2013. Consequently, active and passive measures previously used by Corps contractors to dissuade terns from nesting in the upper estuary were not required.

1.2.2. Columbia Plateau

Methods: Given the relatively small number of Caspian terns nesting at Crescent Island on the mid-Columbia River and Goose Island in Potholes Reservoir (Map 1) compared to the large colony on East Sand Island , estimates of colony size and nesting success at these smaller colonies were based on ground counts of active nests and chicks at the peak of incubation and at the onset of fledging, respectively. Caspian tern colony size, measured as the number of breeding pairs, was based on the maximum number of incubating terns counted on each colony, which is observed late in the incubation period. Nesting success was estimated from the maximum number of fledging-aged birds counted on the colony, which occurs at the beginning of the fledging period. These ground counts were made by researchers from observation blinds situated on the periphery of each tern colony throughout the breeding season (i.e., April – July) and were conducted during daylight hours 2-3 times per week at Crescent Island and 3-4 times a week at Goose Island. No precise measures of variance for our estimates of colony size and nesting success for terns at colonies in the Columbia Plateau region are available. When possible, estimates of the number of breeding pairs from ground counts were verified using counts of nesting adults in oblique aerial photography taken near the peak in incubation period.

Periodic boat-based and aerial surveys of former Caspian tern breeding colony sites in the Columbia Plateau region (i.e., Three Mile Canyon Island, Blalock Islands, Miller Rocks, Cabin Island, Sprague Lake, Banks Lake; Map 1) were conducted during the breeding season to determine whether these colony sites were active. We also flew aerial surveys of the lower and middle Columbia River from Bonneville Dam to Rock Island Dam, the lower Snake River from its mouth to the confluence with the Clearwater River, and Potholes Reservoir searching for new or incipient Caspian tern colonies.

Results and Discussion: Caspian tern attendance at the breeding colony on Crescent Island in 2013 was below the average for 2000-2012 until early June, after which it was slightly higher than the average (Figure 5). A total of 393 breeding pairs of Caspian terns attempted to nest on Crescent Island in 2013, fewer than the number that nested there in 2012 (422 breeding pairs). Caspian tern colony size on Crescent Island trended downward from 2001 to 2007, but has remained relatively stable thereafter (Figure 6).

Both the colony area used by terns (379 m^2 or 0.09 acres) and nesting density (1.04 nests/m^2) declined slightly at the Crescent Island colony in 2013 relative to the previous year (colony area: 397 m^2 or 0.10 acres; nesting density: 1.06 nests/m^2).

We estimated that 169 young terns fledged from the Crescent Island tern colony in 2013, or 0.43 young raised per breeding pair. Nesting success at the Crescent Island Caspian tern colony was below the 14-year average (0.53 young raised per breeding pair) for the 6th consecutive year (Figure 7), possibly due to low availability of juvenile salmonids as prey late in the chick-rearing period (Lyons et al. 2011a).

In 2013, Caspian terns nested on Goose Island in Potholes Reservoir at only one of the two areas previously used for nesting; the main colony, which is located on the western lobe of the island, was the only nesting area for Caspian terns in 2013. The smaller satellite colony, which was located on the smaller eastern lobe of the island, was not occupied in 2013. The peak in colony attendance at the Goose Island colony occurred in mid-May, as in previous years (Figure 8). In 2013, colony attendance at the Goose Island tern colony was generally lower from late April to early June and higher from early June to late July compared to the average weekly attendance observed the previous three years (Figure 8). We estimated that 340 breeding pairs of Caspian terns attempted to nest on Goose Island in 2013, 27% smaller than the estimated colony size in 2012 (463 breeding pairs; Figure 9). Although the Goose Island colony was the largest Caspian tern colony in the Columbia Plateau region during 2009-2012, the Crescent Island tern colony was slightly larger in 2013 (393 breeding pairs). While the colony area used by nesting terns on Goose Island declined in 2013 (319 m^2 or 0.08 acres) compared to 2012 (457 m^2 or 0.11 acres), nesting density increased in 2013 (1.07 nests/m^2) compared to 2012 (1.01 nests/m^2).

We estimated that 130 young fledged from the Goose Island tern colony in 2013, or an average of 0.38 young raised per breeding pair, up from 0.08 young raised per breeding pair in 2012 (Figure 10). In 2010 and 2012, nearly all Caspian tern nesting attempts at Goose Island failed, attributed to a combination of unseasonably cool, wet weather and nocturnal disturbance to nesting terns on the colony by great horned owls (*Bubo virginianus*) and American mink (*Neovison vison*).

Nesting by Caspian terns on the Blalock Island group, located on the mid-Columbia River in John Day Pool, was first detected in 2005 when six pairs attempted to nest on Rock Island. The Rock Island colony peaked at 104 breeding pairs in 2008 and fell to 79 breeding pairs in 2009 before nesting terns abandoned the site and moved to Anvil Island (another island in the Blalock Island group) in 2010 (Figure 11). The tern colony size on Anvil Island declined from its peak in 2010 (136 breeding pairs) to just 6 breeding pairs in 2012. In 2013, nesting Caspian terns relocated to an unnamed island in the Blalock Island group, located between Anvil and Sand islands, where 26 breeding pairs were counted (Figure 11). This new colony only produced 3 fledgling terns, or 0.12 young raised per breeding pair in 2013. This is the eighth consecutive year that Caspian terns nesting at the Blalock Island group have failed or nearly failed to raise any young, either due to nest predation by mammalian or avian predators, or due to high water levels in John Day Pool during the incubation period.

Badger Island, located on the mid-Columbia River in McNary Pool, was home to an incipient Caspian tern colony in 2011 and 2012, where 33 and 60 breeding pairs attempted to nest, respectively. Nesting terns did not return to Badger Island in 2013, perhaps due to complete colony failure the previous two years. Colony failure at Badger Island in 2011 and 2012 was attributed to high water levels in mid-June and/or encroachment and trampling by nesting American white pelicans from the large breeding colony on Badger Island.

In addition to the Caspian tern colony on Goose Island in Potholes Reservoir, we identified two other Caspian tern colonies in the Columbia Plateau region off the Columbia and Snake rivers in 2013. Thirteen pairs of Caspian terns nested on Twinning Island in Banks Lake and one pair nested on Harper Island in Sprague Lake in 2013. From 1997 to 2005, Caspian terns nesting at Banks Lake used Goose Island, north of Twinning Island, where colony size ranged from 10 to 40 breeding pairs. In 2005, Caspian terns began nesting on Twinning Island (also called Dry Falls Dam Island), which is located in Banks Lake just north of Dry Falls Dam. The colony at Twinning Island grew from less than 10 breeding pairs in 2005 to 61 breeding pairs in 2009, before declining to 13 breeding pairs in 2013 (Figure 12). Nesting by Caspian terns on Harper Island in Sprague Lake was first documented in the late 1990's, where they have been nesting sporadically ever since. During 2005-2010, estimates of Caspian tern colony size on Harper Island were generally small (< 10 breeding pairs), before increasing about 6-fold in 2012, and then declining again to just one breeding pair in 2013 (Figure 13). In 2012 and 2013, no young terns were apparently fledged from the colonies at either Twinning Island or Harper Island; the cause[s] of colony failure is unknown. Caspian tern nesting success at Twinning and Harper islands has been generally low, ranging from complete colony failures at both colonies in several years to 0.33 young raised per breeding pair at Twinning Island in 2008 and 2009.

We identified a total of five active Caspian tern colonies in the Columbia Plateau region during 2013 (Figure 14), where a total of approximately 775 breeding pairs nested (Figure 15). The total number of Caspian terns nesting in the Columbia Plateau region in 2013 declined by ca. 20% compared to the average number of breeding pairs counted in the region over the previous 4 years (Figure 15).

1.2.3. Coastal Washington

Methods: Aerial surveys along the southern Washington Coast, Puget Sound, and the Salish Sea, including former and recent Caspian tern colony sites in Willapa Bay, Grays Harbor, Dungeness Spit, Smith Island, the Seattle waterfront, and the Port of Bellingham (Map 1), were conducted on a periodic basis throughout the breeding season in order to detect formation of Caspian tern colonies outside the Columbia River estuary. Survey frequency and methodology did not generally lend themselves to rigorous statistical estimation of measurement uncertainty in colony size or productivity.

The numbers of Caspian terns breeding at sites in the Puget Sound and Salish Sea region of Washington were assessed by a combination of colony counts from oblique aerial photography and periodic ground-based surveys during the breeding season. The number of Caspian terns attempting to nest at Sand Island in Grays Harbor, WA; Dungeness Spit in the Strait of Juan de Fuca, WA; Rat Island in Puget Sound, WA; the rooftops of the Trident Seafood warehouse and the Northwest Industries warehouse in Seattle, WA; the rooftop of the Kimberley-Clark warehouse in Everett, WA; the rooftop of the Fraser River Terminal warehouse in Richmond, British Columbia; and at Smith Island in San Juan National Wildlife Refuge in the Strait of Juan de Fuca (Map 1) were estimated by counting the number of terns attending nests during each visit or by counting apparent attended nests on oblique aerial photography. We also opportunistically assessed nesting chronology, productivity, and factors limiting colony size and nesting success at these colonies throughout the breeding season.

Results and Discussion: Caspian terns were commonly observed foraging and roosting in Willapa Bay and Grays Harbor during the 2013 breeding season. During an aerial survey on 8 June, aerial photography indicated that a new Caspian tern colony had formed on Sand Island in Grays Harbor. In this photography, we counted 133 individual terns, 39 of which were in an incubation posture and one of which was in a brooding posture. During a site visit on 8 July no active nests were observed, and the reason for colony failure is unknown. The available nesting habitat above the high tide line was limited, however, and may have been prone to over-washing during high high tide events. No other nesting attempts were detected in these two southwest Washington estuaries, suggesting that other suitable Caspian tern nesting sites (i.e., islands that include unvegetated substrate above the high high tide level and free of mammalian predators) are not available in Willapa Bay or Grays Harbor.

Based on limited observations, it appeared that Caspian terns did not successfully nest at Dungeness Spit in 2013. Photography taken during an aerial survey 8 June indicated that approximately 110 Caspian terns were on and adjacent to the historical colony site, of which nine appeared to be in an incubation posture. The colony site was abandoned by the last aerial survey on 12 July, when less than 50 terns, all loafing below the high tide line, were observed. Based on these observations, we are confident that any Caspian tern nests that may have been initiated at Dungeness Spit in 2013 failed. Ground surveys were not conducted at Dungeness Spit in 2013 due to the small numbers of terns present during aerial surveys and the time/logistics required for colony site visitation. Nest predation by coyotes (*Canis latrans*) and bald eagles has been a direct cause of nest loss and colony failure in recent years, so it is likely that Caspian tern nesting attempts at this colony site have been negatively affected by repeated colony failures from 2009 to 2012.

The Dungeness Spit Caspian tern colony grew steadily from 2003 to 2009, when it reached ca. 1,500 breeding pairs and was the second largest Caspian tern colony on the Pacific Coast of North America (after the colony on East Sand Island; BRNW 2010a). Based on re-sightings of banded Caspian terns, some growth in the Dungeness Spit tern colony was through immigration of birds from colonies in the Columbia River basin (i.e., East Sand and Crescent islands) and from Commencement Bay, Tacoma, WA (BRNW 2004, 2005b, 2006b, 2009b, 2010b). Despite repeated forays into the Dungeness Spit Caspian tern colony by mammalian predators in previous years, some terns were successful in raising young at the colony in every year until 2009, when coyotes and avian predators caused complete nesting failure for the first time since the colony formed in 2003.

Aerial surveys conducted on 8 June indicated that a small number of Caspian terns may have attempted to nest on Rat Island east of Dungeness Spit in Port Townsend Bay (southeast of Port Townsend). Counts from aerial photography taken during that survey indicated that 24 individuals were on the colony site, and seven appeared to be nesting. During an aerial survey on 12 July, no Caspian terns were observed on the colony site, indicating that any attempted nesting had failed by mid-July. The colony site was limited to a small area between the high tide line and the vegetation, and colony failure may be attributable to over-washing during high high tide events. Additionally, the small colony size may have made it vulnerable to avian predators.

During aerial surveys of the Puget Sound area conducted in 2013, we confirmed that no nesting by Caspian terns occurred at the site of a former Caspian tern colony on the old Georgia-Pacific mill site in the Port of Bellingham, WA for a third straight year. This colony first became established in 2009, when 200 adult terns, some with young, were counted at the site in early July. The colony was located on bare pavement and gravel at the site of a former waterfront warehouse that was demolished and removed in 2008. The area used by nesting terns was surrounded by cyclone fencing, providing some protection from mammalian predators. Our best estimate of colony size in 2010 was between ca. 1,400 and 2,000 breeding pairs. We suspect that some terns that colonized the Port of Bellingham site were from the failed colony at Dungeness Spit, WA; however, re-sightings of previously banded terns at the Port of Bellingham colony indicated that terns also emigrated from colonies in the Columbia River estuary, San Francisco Bay, interior Oregon, and the Columbia Plateau region (see below). Caspian tern productivity at the Port of Bellingham colony was good; we estimated that 900 - 1,400 young terns fledged from the colony in 2010, or an average of 0.5 - 1.0 fledglings per breeding pair. Nest predation, a major limiting factor for colony size and nesting success at other Caspian tern colonies in the region, was not a major factor at this site in 2010. However, due to plans to begin environmental cleanup and development of the site, Caspian terns were actively dissuaded from nesting at the Port of Bellingham site in 2011 and 2012 by employees of the Port of Bellingham.

On the northern-most of three dredge spoil islands in Padilla Bay, a breeding colony of Caspian terns formed in 2011. Estimated colony size was 424 breeding pairs, nesting at three separate areas on the northeast, northwest, and southeast shores of the island. Eggs and chicks were confirmed at this colony, but no young terns were raised to fledging age. Tidal flooding and erosion clearly contributed to breeding failure, but broken eggs and river otter (*Lontra canadensis*) tracks, scat, and a live animal were observed on the colony, suggesting mammalian predation may also have played a role in this colony failure. Observations during aerial surveys showed no evidence that Caspian terns attempted to nest on the Padilla Bay colony in 2012 or 2013; therefore, we did not perform ground-based surveys at Padilla Bay in 2013.

In 2013, an aerial survey conducted on 9 June provided evidence that Caspian terns were congregating in large numbers on Smith Island in the Strait of Juan de Fuca for a third consecutive year. Counts from oblique aerial photography taken on 9 June indicated that there were 609 Caspian terns on the Smith Island colony site, 196 of which appeared to be incubating. By the final BRNW aerial survey on 12 July, the Caspian tern colony had apparently completely failed. The 12 July photography showed that fewer than 50 Caspian terns were on the island, none of which were on the colony site. Caspian terns attempted to nest on Smith Island in both 2011 and 2012, but failed in both years. Nesting habitat for Caspian terns at Smith Island appears to be limited to a small area of bare-sand habitat below the vegetated upland, which is prone to flooding or over-washing during high high tide events. Additionally, a site visit in 2012 indicated that avian predators, including glaucous-winged gulls, bald eagles, and possibly crows were a cause of colony failure that year. Complete colony failure by Caspian terns breeding at Smith Island in 2013 was likely caused by these same factors.

In 2013, Caspian terns attempted to nest on the rooftop of the Trident Seafood warehouse adjacent to Pier 90 in Seattle, WA for the third consecutive year. The colony experienced very low productivity in both 2011 and 2012, with no young fledged in 2011 and 2 - 4 young fledged in 2012. Ground surveys were not conducted by BRNW at this location due to the small numbers of terns present during aerial surveys and the time required for site visitation. Ingrid Taylar with Seattle Audubon provided all ground-based information of this colony in 2013, and first observed birds in an incubation posture on 22 May. On that date, there was a minimum of 16 active Caspian tern nests counted in photography of just a portion of the colony. The colony was apparently abandoned by 27 May. There were night-time disturbances after the terns first arrived at this colony site in April 2013, possibly caused by a great horned owl. A local birder observed a great horned owl in the area during this period and there is a great horned owl nest in Discovery Park, approximately 1 km west of the tern colony. By an aerial survey on 9 June, no Caspian terns were observed on the former colony site, indicating that terns did not successfully nest there in 2013. In 2011 and 2012, we observed intense predation on Caspian tern nests by gulls and crows at this site, particularly during colony flush events. This predation may have contributed to the failure of the colony in 2013 as well.

A second potential rooftop nesting site was located on a large warehouse owned by Northwestern Industries, Inc. in the Magnolia neighborhood of Seattle was reported by Ingrid Taylar. Local birders apparently first noticed Caspian tern activity on this rooftop in 2012. Counts from aerial photography taken during an aerial survey on 9 June indicated that 46 Caspian terns were on the rooftop, and 19 appeared to be in incubation posture. Counts from aerial photography taken during a 12 July aerial survey indicated that the colony size had increased to 133 individuals and 48 active nests; however, the tern nests were not in the same locations as during 9 June survey, indicating that the original nesting attempt had failed and terns had re-nested sometime before the 12 July aerial survey. Ingrid Taylar provided all ground-based information from this site. There was no confirmation of eggs or chicks, and no young were fledged at this site in 2013. Limiting factors for Caspian tern nesting success at this site are unknown, but are likely similar to those of the colony on the Trident Seafoods warehouse.

This was the second year that Caspian terns nested on the roof of the Kimberley-Clark warehouse in Everett, WA. Counts from oblique aerial photography taken on 9 June indicated that there were 1,879 adult Caspian terns on the rooftop, and 1,113 were apparently in incubation posture. There were 90 active nests counted during the first ground-based survey of this colony on 10 June; all active tern nests were located along short linear roof structures where debris had accumulated. The maximum recorded colony attendance on 10 June was 586 terns. Nest numbers decreased steadily throughout the breeding season, down to just 25 active nests and 21 chicks counted on 6 August. From our ground observation location, over 60% of the colony is obscured, however, and our ground-based counts are likely significantly less than the actual numbers of attending terns and active nests at this colony. We counted attended nests in aerial photography taken on 9 June to estimate colony size of the Caspian tern colony on the Kimberly-Clark warehouse at approximately 1,113 breeding pairs in 2013, a significant increase from 2012 (197 breeding pairs). Using oblique aerial photography taken during the 12 July survey, we counted a total of 45 chicks. Using that number, nesting success at the Kimberley-Clark colony was estimated to be at least 0.04 fledglings per breeding pair in 2013. Because only older Caspian tern chicks were visible in aerial photography and because our view is partially obstructed during ground-based observations of this colony, our estimate of colony productivity is likely biased low. During ground-based observation of this colony on 10 June, 66 abandoned eggs (mostly intact) were scattered across the colony, potentially because the concave roof can collect a large amount of water during heavy rainfall events. Three disturbances were observed during a 4-hour monitoring period in early June, but there was no evidence of egg loss due to avian predators during those disturbances.

During an aerial survey on 12 July, a new Caspian tern colony located on the ground 267 m northwest of the Kimberly-Clark warehouse was discovered. Aerial photography taken during that survey indicated that 65 Caspian terns were on the colony site, and 38 appeared to be in incubation posture. In the only ground survey of this site on 6 August, 25 active nests were counted and colony attendance was 67 adult Caspian terns. Some Caspian terns at this new ground colony were sitting tight, presumably on eggs or young chicks, but no eggs or chicks were confirmed. Five fledgling Caspian terns were observed on the colony, but these may have come from the nearby rooftop colony on the Kimberley-Clark warehouse. The new ground colony is highly susceptible to mammalian predators, but no evidence of any disturbance was actually observed.

A new Caspian tern colony was discovered on the rooftop of a warehouse at the Fraser River Terminal in Richmond, British Columbia during 2012 by a local birder, Richard Swanston. Based on limited observations, it appears that terns did not successfully nest at this colony site in 2013. All ground-based observations of this colony were provided by Richard Swanston, and no aerial surveys were conducted at this site due to its proximity to Vancouver International Airport. In 2013, the majority of terns on the warehouse were located on the western half of the rooftop, which is largely obstructed from view from the observation point. Richard estimated a maximum of 400 Caspian terns at the site on 23 June. During a colony flush, no eggs were observed on the eastern half of the rooftop. However, if eggs were present on the western half of the roof, they would have been obscured from view. There were no signs of depredated nests (potential evidence of nest predation by gulls). The colony site was completely abandoned by 22 July.

Loss of former breeding colony sites at Dungeness Spit and the Port of Bellingham has likely contributed to the formation of new Caspian tern colonies at locations in the Salish Sea region over the last three years. A total of 1,300-1,400 nesting attempts by Caspian terns were documented at seven different colonies sites in the Salish Sea region in 2013, but all colonies except the one on the Kimberley-Clark warehouse in Everett failed to raise any young to fledging age. Although the Kimberley-Clark colony experienced low productivity, similar to 2012, the number of breeding pairs grew by at least a factor of five. Continued monitoring in 2014 will be needed to determine if the Kimberley-Clark colony remains active in the future. The overall poor breeding performance by Caspian terns in the Salish Sea region, plus attempts to colonize fenced sites on the mainland and rooftops in urban areas, provide strong support for the hypothesis that suitable nesting habitat for Caspian terns is very limited in the region. Continued monitoring in 2014 and beyond will be necessary to determine where Caspian terns displaced from the former breeding colonies at Dungeness Spit, the Port of Bellingham, and Padilla Bay, plus terns emigrating from the managed colony on East Sand Island, may attempt to nest at new sites in the Salish Sea region.

1.2.4. Interior Oregon and Northeastern California

Methods: Observation blinds were built at the periphery of Caspian tern nesting habitat on each of six Corps-constructed islands that were monitored for Caspian tern nesting activity in interior Oregon (i.e., Crump Lake, East Link impoundment at Summer Lake

Wildlife Area, Gold Dike impoundment at Summer Lake Wildlife Area, Malheur Lake at Malheur National Wildlife Refuge) and in northeastern California (i.e., Sheepy Lake at Lower Klamath National Wildlife Refuge, Tule Lake Sump 1B at Tule Lake National Wildlife Refuge; Map 2). We used a combination of social attraction with tern decoys and audio playback of vocalizations, limited gull control, and continuous monitoring at these recently-constructed islands to help establish and maintain Caspian tern colonies at each site (see Kress 1983 for further details on these methods). Social attraction methods were not used at the Crump Lake tern island in 2013 because managers decided that the Caspian tern colony had exceeded the target number of breeding pairs (500 pairs) in 2009, the last time social attraction was used at that site. Social attraction was not used at the Orems Unit tern island in Lower Klamath NWR because it was not suitable for tern nesting in 2013 (no water was available to fill the impoundment due to drought). Because no nesting attempts by Caspian terns had been detected at the Fern Ridge Reservoir tern island during four years of social attraction at that site (2008-2011), we did not monitor the island for Caspian tern use during the 2012 and 2013 nesting seasons. Data on Caspian tern colony attendance, colony size, productivity, and factors limiting colony size and productivity were collected 1-7 days per week at each of the six monitored islands. Measurement uncertainty in colony size and colony productivity was not expressly estimated; however, repeatability of ground-based counts was generally within 5% or less.

The number of Caspian tern pairs breeding at colonies on Corps-constructed islands in interior Oregon and northeastern California were estimated from ground counts of incubating adult terns near the end of the incubation period. Nesting success (average number of young raised per breeding pair) at each colony was estimated from ground counts of young at the colony at the beginning of the fledging period.

Periodic aerial, road-based, and boat-based surveys of other sites in central, south-central, and southeastern Oregon and northeastern California (Map 4) were conducted during the 2013 nesting season in order to detect nesting attempts by Caspian terns and other piscivorous colonial waterbirds.

Results and Discussion: Caspian terns were observed during the 2013 nesting season at all six of Corps-constructed nesting islands that we monitored in interior Oregon and northeastern California (Figure 16). Caspian terns attempted to nest at five of these islands in 2013 (Crump Lake, East Link, Sheepy Lake, Tule Lake, and Malheur Lake); the Caspian terns observed at the Corps-constructed tern island on Gold Dike did not attempt to nest at that site.

Colony attendance at the Crump Lake tern island in Warner Valley, Oregon during 2013 was similar to what was observed in 2008-2012 until mid-July, when colony attendance dropped precipitously and was far lower in 2013 relative to the average for the previous five years (Figure 17). This difference in late season colony attendance is likely attributable to the earlier onset in fledging at the Crump Lake tern island in 2013 (21

July) compared to the median onset of fledging during 2008-2012 (6 August). About 223 breeding pairs of Caspian terns attempted to nest at the Crump Lake tern colony in 2013, higher than colony size estimates from the previous three years (71, 35, and 115 breeding pairs in 2010, 2011, and 2012, respectively), but down from colony size estimates in 2008 and 2009 (428 and 697 breeding pairs in 2008 and 2009, respectively; Figure 18). As was the case during 2008-2012, high predation rates on Caspian tern eggs were observed at the Crump Lake tern island; California gulls and, to a lesser extent, ring-billed gulls were responsible for the egg predation. High rates of gull predation on tern eggs necessitated the lethal removal of a few problem gulls using firearms (under permit); a total of 12 gulls that were preying on Caspian tern eggs were removed in 2013. We estimated that approximately 61 young Caspian terns fledged from the Crump Lake tern colony in 2013, or an average of 0.27 young fledged per breeding pair (Figure 19).

In 2013, Caspian terns attempted to nest at one of the two Corps-constructed tern islands that remain in the Summer Lake Wildlife Area, the island in East Link impoundment. Although small numbers of Caspian terns (n = 1 – 8 individuals) were observed on the tern island in the Gold Dike impoundment early in the 2013 breeding season (late April through mid-June), no evidence that Caspian terns initiated nesting there was detected, presumably due to frequent disturbances to the colony by great horned owls. Caspian tern colony attendance at the Summer Lake tern islands peaked at 35 adults on-colony in mid-May and declined thereafter (Figure 20). Twenty-one Caspian tern breeding pairs attempted to nest at the East Link tern island in 2013, up from a combined total of 14 breeding pairs nesting at the East Link and Gold Dike tern islands in 2012 (Figure 21). Following two years when no young terns were fledged from either island during 2011-2012, three young terns were fledged from the East Link tern island in 2013, or and average of 0.14 young raised per breeding pair (Figure 22).

Caspian terns attempted to nest at one of the two Corps-constructed tern islands in Lower Klamath NWR, the 0.8-acre floating island at Sheepy Lake. Compared to average Caspian tern colony attendance at the Sheepy Lake tern island during 2010-2012, colony attendance by terns in 2013 was generally higher throughout the breeding season (Figure 23). A total of about 316 breeding pairs attempted to nest at the Sheepy Lake tern island in 2013, the largest colony size so far observed at this site since it was constructed in early 2010 (Figure 24). As was the case at the Crump Lake tern colony, limited gull control was deemed necessary at the Sheepy Lake tern colony; eight gulls that were repeatedly observed depredating tern eggs at the Sheepy Lake colony were shot under permit in 2013. Nesting success of Caspian terns at the Sheepy Lake colony was lower in 2013 (0.37 young raised per breeding pair) compared to the previous year (0.66 young raised per breeding pair; Figure 25). Caspian terns were not observed loafing or attempting to nest on the Orems Unit tern island during the 2013 breeding season, presumably because the Orems Unit impoundment was dry throughout the nesting season.

Colony attendance at the Corps-constructed island in Sump 1B in Tule Lake NWR was higher later in the 2013 breeding season compared to the previous two breeding seasons (Figure 26). The size of the Tule Lake Caspian tern colony was 79 breeding pairs in 2013, higher than the colony size observed when terns first nested there in 2011 (34 breeding pairs), but lower than the colony size observed in 2012 (207 breeding pairs; Figure 27). For the second consecutive year, all Caspian tern nesting attempts on the Tule Lake tern island failed (Figure 28), presumably due to disturbance and nest depredation by a raccoon (*Procyon lotor*) and a great horned owl that repeatedly visited the island at night in both years.

Caspian terns were quick to colonize the Corps-constructed, 1-acre, rock-core island at Malheur Lake in Malheur NWR in 2013, where colony attendance was generally higher in the early and middle parts of the breeding season compared to 2012, the first year when the island was available for tern nesting (Figure 29). Colony size was estimated at 530 breeding pairs in 2013, more than double the estimated colony size the previous year (232 breeding pairs; Figure 30). For the second consecutive year, the Malheur Lake tern island supported the largest Caspian tern colony observed at any of the Corps-constructed tern islands in interior Oregon and northeastern California (Figure 31). Average nesting success at the Malheur Lake tern island in 2013 (0.14 young raised per breeding pair) was considerably lower, however, than the previous year (0.84 young raised per breeding pair; Figure 32). The poor nesting success observed at the Malheur Lake tern island in 2013 was associated with a severe storm event that occurred in early August and caused extensive nest failure, coupled by large numbers (thousands) of American white pelicans that roosted on the island and trampled significant numbers of tern nests.

In 2013, the total number of Caspian terns nesting at Corps-constructed islands created as alternative nesting habitat for Caspian terns displaced from the East Sand Island colony was 1,169 breeding pairs, the highest number recorded since island construction commenced in 2008 (Figure 33). Although predation by gulls and other predators on tern eggs and chicks was the most significant proximal factor limiting the size and productivity of Caspian tern colonies at the Corps-constructed tern islands in interior Oregon and northeastern California during 2013, low forage fish availability was also likely a contributing factor to small colony size and low productivity at some of these sites in 2013 (e.g., Crump Lake tern island and the tern islands at Summer Lake Wildlife Area; Figures 31 and 34). In addition to small colony sizes and low productivity, evidence that food availability is a limiting factor at some of the Corps-constructed islands includes high degree of variation in diet composition from year to year (evidence the food base is inconsistent) and the presences of non-local fishes (e.g., lamprey at Crump Island) in the diet (evidence terns have to travel long distances to forage).

Based on periodic boat-based and aerial surveys, no additional Caspian tern nesting activity were detected at sites outside of the Corps-constructed islands in interior Oregon and northeastern California in 2013. Habitat at the additional colony sites used by Caspian terns in 2011 and 2012 was unsuitable for nesting due to low water levels throughout the region.

Summary: Of the six islands that were monitored for Caspian tern nesting activity, five supported nesting Caspian terns. A combined total of over 1,100 breeding pairs of Caspian terns nested at these five alternative colony sites in 2013, a 50% increase from 2012. Estimated productivity was low among the five sites, however, ranging from an average of 0 to 0.37 young raised/breeding pair, depending on the site. In 2013, mammalian and avian nest predators, displacement by other colonial waterbird species (i.e., California gulls *L. californicus*, American white pelicans), drought, adverse weather condition, and likely low forage fish availability (due to drought), limited Caspian tern colony formation, colony size, and nesting success at one or more of the alternative colony sites. Implementation of various management strategies to minimize the impact of these limiting factors would increase the likelihood of successful colony establishment and productivity (Appendix D, Table D2).

1.3. Diet Composition and Salmonid Consumption

1.3.1. Columbia River Estuary

Methods: Caspian terns transport single whole fish in their bills to their mates (courtship meals) and to their young (chick meals) at the breeding colony. Consequently, taxonomic composition of the diet can be determined by direct observation of adults as they return to the colony with fish (i.e., bill load observations). Observation blinds were set up at the periphery of the tern colony on East Sand Island so that prey items could be identified with the aid of binoculars and spotting scopes. The target sample size was 350 bill load identifications per week. Bill load observations at the East Sand Island tern colony were conducted twice each day, at high tide and at low tide, to control for potential tidal and time of day effects on diet composition. Prey items were identified to the taxonomic level of family. We were confident in our ability to distinguish salmonids from non-salmonids and to distinguish among most non-salmonid taxa based on direct observations from blinds, but we did not attempt to distinguish the various salmonid species. The taxonomic composition of tern diets (percent of identifiable prey items) was calculated for each 2-week period throughout the nesting season. The diet composition of terns over the entire breeding season was based on the average of the percentages for the 2-week periods.

To assess the relative proportion of the various salmonid species in tern diets, we collected fish near the East Sand Island tern colony from Caspian terns returning to the colony with whole fish carried in their bills (referred to hereafter as "collected bill loads"). We employed a non-lethal sampling technique developed in 2011 that utilizes hazing shells to startle terns into dropping their fish; collection of a total of 262 bill load fish was conducted from 22 April to 26 July 2013. No lethal sampling of Caspian terns to determine diet composition was conducted in 2012 or 2013. Salmonid bill loads were identified as either Chinook salmon (*Oncorhynchus tshawytscha),* sockeye salmon (*O. nerka*), coho salmon (*O. kisutch*), steelhead (*O. mykiss*), or unknown based on analyses of morphometrics, diagnostic bones, and genetics[1].

Estimates of total annual smolt consumption by Caspian terns nesting at the East Sand Island colony were calculated using a bioenergetics modeling approach (see Roby et al. [2003] for a detailed description of model structure and input variables). We used a Monte Carlo simulation procedure to calculate reliable 95% confidence intervals for estimates of smolt consumption by Caspian terns.

Results and Discussion: Of the bill load fish identified at the East Sand Island Caspian tern colony during the 2013 nesting season (n = 4,613 bill loads), on average 31% were juvenile salmonids. This proportion of juvenile salmonids in the diet of Caspian terns nesting on East Sand Island, averaged over the entire nesting season, was identical to the 13-year average (31%; Figure 35). As in previous years, marine forage fishes (i.e., anchovies [Engraulidae], surf perch [Embiotocidae], smelt [Osmeridae], and herring [Clupeidae]) were most prevalent, together averaging 63% of all identified bill loads in the diet of terns nesting on East Sand Island in 2013 (Figures 36 and 37). In 2013, the peak in the proportion of salmonids in the diet of Caspian terns nesting on East Sand Island occurred in mid- May, similar to what was observed in previous years based on the weekly averages observed over the previous 13-year period (Figure 38).

Genetic stock identification of salmonid bill load fish collected from East Sand Island Caspian terns in 2011-13 indicated that terns consumed smolts from many of the uniquely identifiable stocks across the basin. For Chinook salmon, the most common genetic stocks of origin for smolts depredated during April and May were the Mid-Columbia River, Upper Columbia River, and Snake River spring run stocks (combined 37 of 57 (77%) Chinook salmon sampled during this period; Figure 39 and Appendix D, Table D3). During June and July, most depredated Chinook salmon smolts (26 of 42 or 62%) originated from the lower Columbia River (Spring Creek Group fall run, West Cascades Tributary fall run, the introduced Rogue River fall run, and the West Cascades Tributary spring run stocks). Depredated steelhead trout originated from six stocks, with steelhead from the Snake River consisting of just over half of the identified samples (53 of 100 samples or 53%; Figure 40 and Appendix D, Table D4). Genetic stock identification was performed for 20 coho salmon smolts collected from terns. The majority of coho originated from the Columbia River stock (17 of 20 samples or 85%; Figure 40 and Appendix D, Table D5).

[1] Genetic analyses were conducted by NOAA Fisheries (POC: David Kuligowski) at the Manchester Field Station genetics laboratory. Species identifications were carried out by amplifying (PCR) the mitochondrial DNA fragment COIII/ND3 as outlined in Purcell et al. (2004). Following species identification, samples were genotyped using species-specific standardized sets of microsatellite DNA markers (Seeb et al. 2007, Blankenship et al. 2011). Stock origins of individual salmon and steelhead were estimated using standard genetic assignment methods (Van Doornik et al. 2007).

Our best estimate of total smolt consumption by Caspian terns nesting on East Sand Island in 2013 was 4.6 million smolts (95% c.i. = 3.9 – 5.3 million smolts), slightly below the average of the previous 13 years for the third consecutive year (Figure 41). From 2000 to 2012, the average number of smolts consumed by Caspian terns nesting on East Sand Island was 5.3 million smolts per year (Figure 41). This is less than half the annual consumption of juvenile salmonids by Caspian terns nesting in the Columbia River estuary prior to 2000, when the breeding colony was located on Rice Island in the upper Columbia River estuary.

Of the juvenile salmonids consumed by East Sand Island Caspian terns in 2013, we estimate that 1.7 million or 37% were coho salmon (95% c.i. = 1.5 – 2.0 million smolts), 0.9 million or 19% were steelhead (95% c.i. = 0.7 – 1.0 million smolts), 1.1 million or 24% were sub-yearling Chinook salmon (95% c.i. = 1.0 – 1.3 million smolts), 0.9 million or 19% were yearling Chinook salmon (95% c.i. = 0.7 – 1.0 million smolts), and 0.02 million or < 1% were sockeye salmon (95% c.i. = 0.02 – 0.03 million smolts; Figure 42).

1.3.2. Columbia Plateau

Methods: The taxonomic composition of the diet of Caspian terns nesting on Goose Island in Potholes Reservoir was determined by direct observation of adults as they returned to the colony with fish (i.e., bill load observations; described above). The target sample size at Goose Island was 150 bill load identifications per week during the peak of the breeding season (see above for further details on the analysis of diet composition data). Prey items were identified to the taxonomic level of family. We identified prey to species, where possible, and salmonids were identified as steelhead trout or 'other salmonids' (i.e., Chinook salmon, coho salmon, or sockeye salmon). Trout were distinguished from 'other salmonids' by the shape of the caudal fin, body shape, coloration and speckling patterns, shape of parr marks, or a combination of these characteristics (see Antolos et al. 2005). The percent of identifiable prey items in tern diets was calculated for each 2-week period throughout the nesting season. The diet composition of terns over the entire breeding season was based on the average of the percentages from these 2-week periods. Bill load fish were not collected nor were terns hazed at the Goose Island tern colony in order to assess diet composition because of the potential impact of lethal or non-lethal diet sampling on such a small breeding colony. Diet composition data was not collected at the Crescent Island Caspian tern colony during 2013.

An estimate of annual smolt consumption by Caspian terns nesting at the Goose Island colony was calculated using a bioenergetics modeling approach (see Antolos et al. [2005] for a detailed description of model structure and input variables). Both steelhead smolts from the Columbia River and resident rainbow trout stocked in Potholes Reservoir (and other nearby water bodies) were available to Goose Island terns. Based on the morphology (degree of smoltification) of each identified fish, it was possible to confidently classify 94% of the *O. mykiss* brought to the colony as steelhead smolts, leaving 6% as unidentified, either steelhead or resident rainbow trout. This uncertainty in the identification of bill load fish caused us to calculate consumption estimates based on two different scenarios. First, we assumed that all *O. mykiss* identified in tern bill loads were anadromous steelhead smolts from the upper Columbia River (upper bound of the estimate of anadromous salmonid consumption by terns) and, second, we assumed that 94% were steelhead and the remainder were resident rainbow trout (lower bound of the estimate of anadromous salmonid consumption by terns).

Results and Discussion: Of the bill load fish identified at the Caspian tern colony on Goose Island in Potholes Reservoir, an average of 10% were juvenile salmonids (n = 1,837 identified bill loads), a 3-fold decline as compared to the previous year (30%) and the lowest percentage of salmonids in the diet ever observed at that colony (Figure 43). Based on morphological characteristics of the salmonids identified at the colony, we estimate that a minimum of 98% of the identified salmonids were anadromous fish (steelhead or salmon) from the Columbia River, with some portion of the remainder being resident trout from Potholes Reservoir and perhaps other nearby lakes and reservoirs. The fact that Caspian terns commute over 100 km round trip from the nesting colony to the Columbia and Snake rivers to forage (see Appendix C; Maranto et al. 2010), suggests that adult terns provisioning themselves may sometimes find more profitable foraging opportunities in these distant river systems than in Potholes Reservoir and closely surrounding water bodies. Given that Caspian terns foraging in the Columbia River or Snake River conduct fewer foraging trips per day than those terns foraging in and near Potholes Reservoir, they have fewer opportunities to return to the colony carrying a fish (e.g., anadromous salmonids) from the rivers. It is likely that anadromous salmonids are under-represented in the Goose Island tern diet data, which is based on observations of fish carried by terns back to the colony. This under-representation of salmonids in the diet in 2013 may have been more pronounced than in recent years if terns foraging near the colony had greater access to prey and were able to make shorter and more frequent foraging trips to provision mates and chicks. The proportion of juvenile salmonids in bill loads of Goose Island Caspian terns was highest in mid-May (29% of identifiable bill loads) during 2013, a week earlier than the peak in the proportion of salmonids in the diet during the previous year (83% of identifiable bill loads) (Figure 45).

Because few *O*. *mykiss* were not identified as either anadromous steelhead trout or resident rainbow trout, estimates of total smolt consumption were similar for scenarios assuming that unidentified trout were entirely steelhead or entirely rainbow trout. Assuming all unidentified trout were resident rainbow trout, we estimated that Caspian terns nesting on Goose Island consumed 57,000 anadromous juvenile salmonids (95% c.i. = 48,000 – 65,000 smolts). Assuming all unidentified *O. mykiss* were steelhead, we estimated that 58,000 anadromous juvenile salmonids were consumed (95% c.i. = 49,000 – 67,000 smolts; Figure 46). Based on the species composition of Caspian tern bill loads identified on the colony at Goose Island-Potholes, and assuming all unidentified trout were steelhead, we estimate that salmon smolts (i.e., Chinook, coho, or sockeye) comprised 66% of the estimated total number of anadromous salmonid smolts consumed by Goose Island terns in 2013, or about 37,000 smolts (95% c.i. = 32,000 – 44,000 smolts). Steelhead comprised 33% of all juvenile salmonids consumed or approximately 19,000 smolts (95% c.i. = 16,000 – 22,000 smolts; Figure 47). Estimates of predation rates on steelhead smolts by Goose Island Caspian terns based on smolt PIT tag recoveries on-colony indicate that steelhead consumption in 2013 was greater than 19,000 smolts (see section 1.4.2). Similar discrepancies between steelhead predation rates based on PIT tag recoveries and steelhead consumption estimates based on bioenergetics modeling were detected each year during 2010-12. This discrepancy is consistent with an under-representation of salmonids in the Goose Island tern diet data, resulting from differential provisioning of mates and chicks by terns foraging at the Columbia and Snake rivers versus in Potholes Reservoir and other locations near the colony. Consequently, bioenergetics estimates of juvenile salmonid consumption by Goose Island Caspian terns that are based on diet observations at the colony should be interpreted as minimum estimates of smolt consumption.

1.3.3. Coastal Washington

No diet composition data were collected for Caspian terns nesting along the Washington coast in 2013.

1.3.4. Interior Oregon and Northeastern California

Methods: The taxonomic composition of the diet of Caspian terns nesting on the Corps-constructed tern islands at Crump Lake, East Link, Sheepy Lake, Tule Lake Sump 1B, and Malheur Lake were determined by direct observation of adults as they returned to the colony with fish (i.e., bill load observations; described above). Bill load fish we identified each week throughout the breeding season at each colony site (see above for further details on the analysis of diet composition data). Fish were identified to the lowest taxonomic grouping possible using visual observation. Visual identifications were verified using voucher specimens, whenever possible. Breakdown of diet composition in identified samples is provided below; measurement uncertainty was not estimated. In addition to the visual identification of fish, PIT tags were recovered on selected tern colonies to estimate tern predation rates on fish species of special concern to resource managers (e.g., Warner suckers [*Catostomus warnerensis*] at Crump Lake tern island and Lost River suckers [*Deltistes luxatus*] and shortnose suckers [*Chasmistes brevirostris*] at Sheepy Lake and Tule Lake Sump 1B tern islands; see Section 1.4.4.).

Results and Discussion: A moderate number of Caspian tern bill loads (n = 1,544) were identified at the Crump Lake Caspian tern colony in 2013. The diet composition of Caspian terns nesting on the Crump Lake tern island in 2013 consisted primarily of cyprinids (chub, minnows, and carp; 46% of identifiable prey items), centrarchids (crappie, sunfish, and bass; 30%), followed by ictalurids (catfish; 22%; Figure 48). Diet composition at the Crump Lake tern colony in 2013 was similar to 2008-2009 when cyprinids (primarily Tui chub [*Gila bicolor*]) dominated the diet. In 2010-2012, the diet of terns nesting at the Crump Lake tern island was dominated by centrarchids (primarily white crappie [*Pomoxis annularis*]). During 2010-2013, only one juvenile sucker was observed among the identified prey items at the Crump Lake tern island, and that sucker was observed in 2012. Six suckers were observed in the bill loads of Caspian terns at the Crump Lake colony during 2008-2009, five in 2008 alone (< 0.1% of identifiable prey items). Of all seven suckers identified in the diet of Caspian terns at the Crump Lake tern island during 2008-2013, only one could be positively identified as an ESA-listed Warner sucker, and it was observed in 2008 (see Section 1.4.4). It is unknown whether the other 6 suckers were ESA-listed Warner suckers or an unlisted species of sucker.

A small number of Caspian tern bill loads (n = 130) were identified at the East Link Caspian tern colony in Summer Lake Wildlife Area during 2013. As was the case in 2009-2012, the diet composition of Caspian terns nesting at Summer Lake Wildlife Area in 2013 was dominated by cyprinids (primarily Tui chub; 95% of identifiable prey items; Figure 49). In 2013, rainbow trout comprised only 2% of the diet of Caspian terns nesting at the Summer Lake Wildlife Area, compared to 13% during 2009-2012 (Figure 49). Based on fish watch observations, suckers were not detected in the diet of Caspian terns nesting at Summer Lake Wildlife Area in 2011-2013. One sucker (0.3% of identifiable prey items) was observed among the identified prey items at the East Link tern colony in 2010. It is unknown whether this sucker was an ESA-listed Warner sucker or an unlisted species. Warner suckers are not endemic to Summer Lake, although a small number of Warner suckers were intentionally relocated to the area by the Oregon Department of Fish and Wildlife and the U.S. Fish and Wildlife Service several years ago as part of a salvage operation due to drought conditions in the Warner Valley (P. Scheerer, ODFW, pers. comm.).

A moderate number of Caspian tern bill loads (n = 1,471) were identified at the Sheepy Lake colony in 2013. The diet composition of Caspian terns nesting on the Sheepy Lake tern island was dominated by cyprinids (primarily chub and fathead minnows [*Pimephales promelas*]), at 85% of identifiable prey items), followed by centrarchids (primarily Sacramento perch [*Archoplites interruptus*]), at 10% of identifiable prey items (Figure 50). No juvenile suckers were detected in the diet of Caspian terns nesting at the Sheepy Lake colony in 2013, based on bill load identifications and sucker PIT tag recoveries (see Section 1.4.4). One juvenile sucker (< 0.1% of identifiable prey items) was observed among the identifiable prey items at the Sheepy Lake tern colony in 2011. The sucker seen at the Sheepy Lake tern colony could not be positively identified as either an ESA-listed Lost River sucker or an ESA-listed shortnose sucker. An un-listed species of sucker, the Klamath largescale sucker (*Catostomus snyderi*), is also found within foraging distance of the Sheepy Lake tern island.

A total of 978 Caspian tern bill loads were identified at the Tule Lake Sump 1B colony in 2013. The diet composition of Caspian terns nesting on the Tule Lake Sump 1B tern island was dominated by cyprinids (primarily chub and fathead minnows), at 75% of identifiable prey items, followed by centrarchids (primarily Sacramento perch), at 24% of identifiable prey items (Figure 51). Based on bill load identifications and PIT tag recoveries (see Section 1.4.4), juvenile suckers were not detected in the diet of Caspian terns nesting at the Tule Lake Sump 1B colony in 2013. One juvenile sucker (< 0.1% of identifiable prey items) was observed among the identifiable prey items at the Tule Lake Sump 1B tern colony in 2011. As was the case at the Sheepy Lake tern colony, this sucker could not be identified as either an ESA-listed or unlisted sucker species.

A large number of Caspian tern bill loads (n = 2,400) were identified at the Malheur Lake colony in 2013. The diet composition of Caspian terns nesting on the Malheur Lake tern island was dominated by cyprinids (primarily common carp [*Cyprinus carpio*], at 90% of the identifiable prey items), followed by ictalurids (catfish), at 7% of identifiable prey items (Figure 52). Fifteen trout (0.5% of identifiable prey items) were identified among the bill loads at the Malheur Lake tern island. Based on the size (16 – 25 cm total length) and capture dates (May through early July) of these trout, most, if not all, were likely rainbow trout stocked in nearby lakes and reservoirs.

1.4. Predation Rates Based on Salmonid Smolt PIT Tag Recoveries

Passive integrated transponder (PIT) tags are placed in salmonids and other fishes to study their behavior and survival following tagging and release. Salmonid smolt PIT tags were first discovered on piscivorous waterbird colonies in the Columbia River basin during 1996 (Collis et al. 2001). Beginning in 1998, specially-designed electronics (antennas and transceivers) were developed and used to recover PIT tags *in situ* on bird colonies in the Columbia River basin (Ryan et al. 2001). PIT tags provide specific information on each tagged fish, including species, rear-type (hatchery or wild), run-timing, and temporal availability (based on detections of live fish passing PIT tag antenna arrays during out-migration). Recoveries of PIT tags on piscivorous bird colonies can be used to estimate predation rates and to compare the relative susceptibility of different fish populations to avian predation (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Frechette et al. 2012, Sebring et al. 2013).

The main objectives for using information collected from smolt PIT tags for this study were to (1) determine colony-specific avian predation rates on particular salmonid ESUs (2) assess differences in predation rates on smolts based on bird species and location of bird nesting colonies, and (3) evaluate whether avian predation rates in 2013 were similar to those reported in previous years. Comparisons between current and historical predation rates were made in the context of on-going or proposed management initiatives for piscivorous colonial waterbirds as a means of evaluating those initiatives in reducing avian predation on salmonid populations and other fish of conservation concern (e.g., ESA-listed catostomids).

Research aimed at recovering PIT tags from bird colonies in the Columbia River estuary was conducted in collaboration with NOAA Fisheries (POC: Jen Zamon). Research in interior Oregon and northeastern California was conducted in collaboration with the Oregon Department of Fish and Wildlife (POC: Ben Ramirez), the USGS-Klamath Falls Field Station (POC: Dave Hewitt), and the USFWS-Upper Klamath Basin National Wildlife Refuges (POCs: Dave Mauser and John Beckstrand), and focused on avian predation on ESA-listed sucker species and redband trout by Caspian terns nesting at the Corps-constructed alternative colony sites located on Crump, Sheepy, Tule, and Malheur lakes.

Methods: Quantification of avian predation rates involves recovery of fish tags on bird colonies (Collis et al. 2001; Ryan et al. 2003; Evans et al. 2011b; Evans et al. 2012; Frechette et al. 2012). Accurate estimation of avian predation rates based on fish tags recovered on birds colonies incorporates several important metrics, including (1) the number of tagged fish available (e.g., released or interrogated in the vicinity of a bird colony), (2) the number of available tagged fish recovered on the bird colony, (3) estimate(s) of detection efficiency (i.e., probability of detecting a tag if it was deposited on the colony), and (4) on-colony deposition rate estimates (i.e., probability a consumed tag was deposited on-colony).

Fish Availability - We queried the regional salmonid PIT Tag Information System database (PTAGIS 2013), maintained by the Pacific States Marine Fisheries Commission, to acquire data on PIT-tagged smolts released and interrogated passing dams in the Columbia River basin during 2013. Following the methods of Evans et al. (2012), PIT-tagged smolts were grouped by evolutionarily significant unit (ESU) or distinct population segment (DPS) of anadromous salmonid, with each ESU/DPS representing a unique combination of species (Chinook salmon, coho salmon, sockeye salmon, or steelhead), run-type (spring, summer, fall, or winter), and river-of-origin (Columbia, Snake, or Willamette). The designation of ESUs/DPSs follows that of NOAA (2013), which includes both wild and hatchery-reared fish. All PIT-tagged salmonids of the appropriate species and run-type that were tagged and released within the geographic boundary of the NOAA-defined ESU/DPS were included in the study, as long as the fish was interrogated passing a dam upstream of the bird colony of interest, following the methods of Evans et al. (2012).

PIT Tag Recovery - The methods described in Evans et al. (2012) and Zamon et al. (2013) were used to recover PIT tags from bird colonies in the Columbia River basin in 2013. PIT tag antennas were used to recover PIT tags *in situ* during August through November, after birds dispersed from their breeding colonies. PIT tags were detected by systematically scanning the entire area occupied by birds during the nesting season (referred to as a "pass"), with a minimum of two passes or complete sweeps of the nesting area made at each bird colony. The area occupied by birds on each colony was determined from aerial photography of the colony and visits to the colony during the nesting season.

PIT Tag Detection Efficiency - Not all PIT tags deposited by birds on their nesting colony are subsequently found by researchers after the nesting season. PIT tags can be blown off the colony during wind storms, washed away during high tides, rain storms, or other flooding events, or otherwise damaged or lost during the course of the nesting season. Furthermore, the detection methods used to find PIT tags on bird colonies are not 100% efficient, with some proportion of detectable tags missed by researchers during the scanning process. To address these factors, PIT tags with known tag codes were intentionally sown on the colony (hereafter referred to as "control tags") throughout the nesting season at each bird colony to quantity PIT tag detection efficiency. The sowing of control tags was conducted during two to four discrete periods during the birds' nesting season: (1) prior to arrival of birds (March to April), (2) during the egg incubation period (April to May), (3) during the chick-rearing period (May to June), and (4) immediately following the fledging of young (July to September). These periods were selected because they encompassed the time periods when juvenile salmonids were out-migrating and therefore available as prey to nesting birds. The total number of control PIT tags sown varied by colony, with sample sizes ranging from 150 PIT tags to 400 PIT tags per colony. The number of discrete time periods when control tags were sown also varied, but was no less than two (at the beginning and end of the nesting season) and no more than four. During each release, control tags were haphazardly sown throughout the entire area occupied by nesting birds during the breeding season.

Deposition Rates - Not all smolt PIT tags that are ingested by breeding birds are subsequently deposited on their nesting colony. A portion of the PIT tags implanted in depredated fish are stolen by other predators (kleptoparasitized), damaged and rendered unreadable during digestion, or are excreted off-colony at loafing, staging, or other areas utilized by birds during the nesting season. The proportion of ingested PIT tags that are subsequently deposited intact on the breeding colony is referred to as PIT tag "deposition rates". Methods and results from studies aimed at quantifying PIT tag deposition rates for nesting Caspian terns, double-crested cormorants, and California gulls in 2013 are presented in Appendix A (Deposition Studies). Briefly, trout with known PIT tag codes were fed to nesting birds and the number of tags found by researchers on the colony at the end of the nesting season was used to estimate an on-colony PIT tag deposition rate. Data on detection efficiencies and deposition rates were then used estimate predation rates for each tern colony (see below).

Predation Rate Calculations - Predation rates for each available PIT-tagged salmonid ESU were calculated using an iterative multi-step approach. First, logistic regression was used to interpolate colony specific daily detection efficiencies, whereby a binary response of control tag detections (detected or not detected) was modeled as a function of time since control tags were sown on the bird colony (eq. 1).

$$(1)\quad \widehat{p_j} = \frac{e^{(\beta_0 + \beta_1 t_j)}}{1 + e^{(\beta_0 + \beta_1 t_j)}}$$

where $\widehat{p_j}$ is the probability of detecting a tag deposited on day *j*, β_0 is the regression intercept, β_1 is the regression slope, and t_j is the independent variable for date *j*. The number of fish consumed on day *j* can be estimated as the number of fish recovered on day *j* (r_j) divided by the estimated the detection efficiency on day *j* ($\widehat{p}_j$) and the on-colony deposition rate (ϕ; eq. 2).

$$(2)\quad \widehat{c}_j = {r_j}/{(\widehat{p_j} * \widehat{\phi})}$$

On-colony deposition rate ($\widehat{\phi}$) was estimated from previous studies (Caspian terns) and on-going studies (gulls and double-crested cormorants) described in Appendix A. Annual predation rates were estimated by dividing the estimated number of consumed fish by the number of fish released across an entire season. These calculations were conducted independently for each bird colony and salmonid ESU of interest.

Confidence intervals for ESU-specific predation rates were estimated by a bootstrapping simulation technique (Efron & Tibshirani 1986; Manly 1998). The bootstrapping analysis consisted of 2,000 iterations of all calculations, with each iteration representing a unique bootstrap re-sample (random sample with replacement) of all datasets: detection efficiency, on-colony deposition, fish availability, and tag recoveries. The 2.5th and 97.5th quartiles were used to represent the limits of a bootstrapped 95% predation rate confidence interval.

To control for imprecise results that might arise from small sample sizes of interrogated PIT-tagged smolts, estimates of predation rates were only calculated for ESUs/DPSs when ≥ 500 PIT-tagged salmonids were interrogated passing an upstream dam in a given year. Predation rates < 0.1% are presented without confidence intervals because of the proximity of the estimate to zero. Additionally, only PIT-tagged smolts detected at a dam during the bird nesting season (1 March to 31 August, depending on the colony) were included in these analyses, as these salmonids were believed to be available to birds nesting at the colony. Analyses were conducted using R statistical software, with statistical significance set at $\alpha = 0.05$.

Results from this modeling procedure for estimating avian predation rates on PIT-tagged salmonid smolts were based on the following assumptions (see Evans et al. 2012 for additional information):

- A1. Salmonid smolt release and interrogation information obtained from PTAGIS was complete and accurate.
- A2. PIT-tagged smolts detected passing an upstream dam were available to avian predators nesting downstream of that dam.
- A3. The detection probability for control PIT tags sown on-colony is equal to that of PIT tags naturally deposited by birds.
- A4. PIT tag deposition rates obtained from study fish fed to birds were equal to that of PIT-tagged smolts naturally consumed and deposited by birds.

A5. PIT tags from consumed fish were deposited on a bird colony within a short time period (days, week) of the fish being detected passing an upstream dam.

A6. PIT-tagged fish, by species, ESU, rear-type, and detection site (dam), were representative of non-tagged fish passing the same dam.

To verify the first assumption (A1), irregular entries were either validated by the respective coordinator of the PIT-tagging effort or eliminated from the analysis. Detections of PIT-tagged salmonids at dams upstream of bird colonies were deemed the most appropriate measure of fish availability given the downstream movement of juvenile salmonids, the ability to standardize data across all sites, and the ability to define unique groups of salmonids by a known location and passage date (Assumption A2). Detection efficiency estimates (A3) were generally high at all colonies (see Results); thus, possible violations of assumption A3 would have little effect on estimates of predation rates. At this time there are no data available to support or refute assumption A4, other than to note that PIT tag deposition rates are known to vary by avian predator (tern, cormorant, gull) and that this variation was incorporated into predation rate estimates (see Appendix A). Assumption A5 relates to the use of the last date of live detection as a proxy for the date a PIT tag was deposited on a bird colony, and needs to be only roughly true because detection efficiency did not change dramatically on a daily basis (see Results). Assumption A6 relates to interference regarding the consumption of PIT-tagged fish and all fish (tagged and untagged) susceptible to avian predation. Similar to A4, there are few empirical data to support or refute assumption A6, other than to note that the run-timing and abundance of PIT-tagged fish is often in agreement with the run-timing and abundance of non-tagged fish passing dams on the Columbia and Willamette rivers.

1.4.1 Columbia River Estuary

Methods: Salmonid impacts based on PIT tag recoveries at the Caspian tern colony on East Sand Island were evaluated in 2013 using methods described in Section 1.4.

Results and Discussion: Following the nesting season, 11,830 PIT-tagged smolts from the 2013 migration year (Chinook salmon, coho salmon, sockeye salmon, and steelhead smolts combined from all release sites) were recovered on the East Sand Island Caspian tern colony (Table 2). Recoveries of control tags sown on the East Sand Island tern colony (n = 300) indicated that detection efficiency ranged from 41% to 72% for tags deposited between 1 March and 31 August (Table 3 and Appendix D, Figure D1). Deposition rates for East Sand Island Caspian terns were estimated to be 71% (95% c.i. = 62-81%; Table 4 and Appendix A).

Based on predation rates of PIT-tagged smolts last detected passing Bonneville Dam (lower-most dam on the Columbia River) or Sullivan Dam (lower-most dam on the Willamette River; Map 1), steelhead were the most susceptible salmonid species to predation by Caspian terns nesting on East Sand Island in 2013; predation rates on steelhead DPSs ranged from 8.6% (95% c.i. = 7.1-10.6%) on Upper Columbia River steelhead to 12.5% (95% c.i. = 10.4-15.1%) on Snake River steelhead (Table 5). Predation rates on Chinook salmon ESUs were significantly lower than those of steelhead, ranging from 0.6% (95% c.i. = 0.2-1.2%) on Upper Columbia River spring Chinook to 1.4% (95% c.i. = 0.8-2.0%) on Upper Columbia River Summer/Fall Chinook. Similar to Chinook ESUs, predation on Snake River sockeye salmon (0.7%; 95% c.i. = 0.2-1.5%) was significantly lower compared to steelhead DPSs (Table 5).

Of those ESUs with adequate sample sizes of PIT-tagged smolts, there was no significant difference in tern predation rates on fish based on rearing type, whereby predation rates on wild fish were similar – albeit slightly lower – to those on hatchery fish of the same ESU/DPS (Table 6). Evans et al. (2011a) also found no significant difference in the relative susceptibility of hatchery and wild fish to East Sand Island Caspian tern predation during 2004-2009. Differences in predation rates between hatchery and wild fish, however, have been consistently reported at other avian colonies in the basin (e.g., Goose Island terns and Table 6).

In general, predation rates on salmonid smolts by East Sand Island Caspian terns in 2013 were similar to those observed during 2011 and 2012 (Appendix B), but were generally lower than estimated predation rates during 2007-2010 (Appendix B). For example, average annual predation rates on Snake River steelhead during 2007-2010 were 16.4% (range = 14.0-22.6%), but averaged 11.5% (range = 10.0-12.8%) during 2011-2013. Similar reductions in tern predation rates have been observed in other steelhead and salmon DPSs/ESUs (BRNW 2013a). General reductions in ESU-specific predation rates during 2011-2013 coincided with comparable reductions in colony size (Figure 2) and consumption estimates (Figure 41) relative to during 2007-2010. This suggests that Caspian tern management initiatives at the East Sand Island colony to reduce nesting habitat are beginning to result in reductions in tern predation rates on salmonids. It is worth noting, however, that there was a slight increase in colony size in 2013 (ca. 7,100 pairs) compared to 2012 (ca. 6,400 pairs) and predation rates on steelhead stocks were generally higher in 2013 but were still lower than those observed during 2007-2010. Apparent management-derived reductions in tern predation rates on salmon ESUs (those originating above Bonneville and Sullivan dams) were not, however, negatively influence by the slightly larger tern colony in 2013 compared to 2012 (i.e., increases in 2013 were germane to steelhead DPSs only).

Adequate sample sizes (≥ 500 interrogated PIT-tagged smolts) were not available for all ESUs/DPSs originating entirely above Bonneville or Sullivan dams in 2013. For example, there were < 200 known-origin sockeye salmon, Chinook salmon, or winter-run steelhead PIT-tagged and released within the geographic range of the Lake Wenatchee, Okanogan River, Deschutes River, or upper Willamette River ESUs/DPSs that were subsequently detected at Bonneville Dam or Sullivan Dam in 2013. It should also be noted that data regarding the impacts of East Sand Island Caspian terns on survival of

PIT-tagged smolts originating from lower Columbia River ESUs/DPSs are not presented here due to the paucity of in-stream PIT tag detectors below Bonneville Dam and an insufficient sample size of released PIT-tagged fish. As such, the impacts of predation by East Sand Island Caspian terns on Lower Columbia River salmonid populations, some of which are ESA-listed (i.e., Chinook salmon, coho salmon, steelhead), are largely unknown and require a different analytical framework to evaluate (see Lyons et al. 2012).

1.4.2 Columbia Plateau

Methods: An evaluation of impacts of Caspian tern predation to smolt survival based on PIT tag recoveries at Caspian tern colonies in the Columbia Plateau region were conducted at three sites in 2013: (1) Goose Island in Potholes Reservoir, WA, (2) Crescent Island on the mainstem Columbia River (McNary Pool), and (2) the Blalock Islands on the mainstem Columbia River (John Day Pool). The methods for calculating predation rates on juvenile salmonids based on PIT tag recoveries at these three Caspian tern colonies are the same as those described in Section 1.4, except that the availability of juvenile salmonids to tern predation was based on detections of PIT-tagged fish at Rock Island Dam on the middle Columbia River or Lower Monumental Dam on the lower Snake River for the terns nesting at Goose Island and Crescent Island, and at McNary Dam on the Columbia River for terns nesting on the Blalock Islands.

To supplement the small numbers of ESA-listed PIT-tagged fish annually interrogated passing Rock Island Dam on the middle Columbia River, steelhead and yearling Chinook were captured, PIT-tagged, and released into the tailrace of Rock Island Dam from 10 April to 15 June 2013. Tagging was part of research funded by Grant County Public Utility District No. 2. Steelhead and yearling Chinook were randomly-selected for tagging at the dam and were tagged in concert with, and in proportion to, the run at-large passing Rock Island Dam. In addition to PIT-tagging, fish were measured (mm, fork length), weighed (g), condition-scored, placed in a recovery tank, and then released into the tailrace of the dam to resume out-migration. To reduce handling time, high-resolution digital photography was taken of each side of the fish, which allowed for a detailed classification of the external condition of the fish by type and magnitude. We assessed the incidence and severity of several different anomalies (body injuries, descaling, signs of disease, fin damage, and ectoparasitic infestation) for each tagged fish using the methods of Hostetter et al. (2011). This condition assessment allowed us to determine whether fish with noticeable external damage and disease were more susceptible to avian predation than seemly healthy fish (see Hostetter et al. 2011 and Hostetter et al. 2012 for additional details regarding these methods).

Recoveries of smolt PIT tags on the Caspian tern colony at Goose Island in Potholes Reservoir were used to determine predation rates and to assess whether susceptibility to avian predation varied by individual fish characteristics (e.g., condition and fork length). Recoveries of PIT tags on the Crescent Island and Blalock Islands tern colonies were only used to determine predation rates because all smolts last detected at Lower Monumental Dam or McNary Dam were not condition-scored at these facilities. As previously noted (Section 1.4), predation rates at all tern colonies included in the study in 2013 were adjusted to account for PIT tag detection efficiency rates (Table 3 and Appendix D, Figure D1) and on-colony PIT tag deposition rates (Table 4 and Appendix A).

Results and Discussion:

Goose Island Caspian terns – Following the nesting season, a total of 2,676 PIT-tagged smolts from the 2013 migration year (Chinook, coho, sockeye, and steelhead combined from all releases) were recovered on the Goose Island Caspian tern colony (Table 2). Control tags sown on the Goose Island tern colony (n = 400) indicated that detection efficiency ranged from 22% to 90% for tags deposited between 1 April and 31 July (Table 3 and Appendix D, Figure D1). Deposition rates for Goose Island Caspian terns were estimated to be 71% (95% c.i. = 62-81%; Table 4 and Appendix A).

As part of our fish tagging efforts at Rock Island Dam, a total of 5,893 steelhead smolts (4,284 hatchery, 1,609 wild) and 5,759 yearling Chinook smolts (5,759 hatchery, 277 wild) were PIT-tagged, condition-scored, and released from 10 April to 15 June 2013. All (100%) of the steelhead tagged at Rock Island Dam were part of the ESA-listed Upper Columbia River steelhead DPS. Not all of the yearling Chinook tagged at Rock Island Dam, however, were part of the ESA-listed spring-run ESU because non-listed summer-run hatchery Chinook are also released as yearlings above Rock Island Dam. Based on unique markings (a combination of fin clips and coded wire tags), a minimum of 992 of the yearling Chinook smolts tagged at Rock Island Dam were considered ESA-listed spring-run Chinook, while the remaining fish were a mixture of hatchery spring-run and hatchery summer-run Chinook.

Fish sampling numbers at Rock Island Dam peaked in May, with 80% of all steelhead sampled between 9 May and 28 May and 80% of all yearling Chinook sampled between 30 April and 28 May. Condition sampling indicated that 10.6% and 4.9% of steelhead and yearling Chinook, respectively, had moderate-to-severe external signs of damage and disease (i.e., open body wounds, > 25% descaling, fungal infections, or other anomalies; see Hostetter et al. 2011).

Of the steelhead smolts tagged and released into the tailrace of Rock Island Dam, an estimated 14.9% (95% c.i. = 12.7-17.8) were consumed by Caspian tern nesting on Goose Island in Potholes Reservoir in 2013 (Table 7). The estimated predation rate by Caspian terns nesting at Goose Island in 2013 was lower than the estimate in 2012 (17.3%), but close to the average (15.3%) during 2008-2012 (Appendix B). Data collected to date indicate that there is a positive relationship between annual Caspian tern predation rates on juvenile steelhead and the number of Caspian terns breeding at Goose Island, with predation rates higher in years when colony size was greater (Figure 53).

Of the yearling Chinook salmon smolts tagged and released into the tailrace of Rock Island Dam, an estimated 1.7% (95% c.i. = 1.1-2.4) were consumed by Caspian terns nesting on Goose Island in 2013. Of the known ESA-listed spring-run Chinook (n = 992), an estimated 2.1% (95% c.i. = 0.7- 4.0) were consumed by Caspian tern nesting on Goose Island in 2013 (Table 7). There was no statistical difference in tern predation rates on the known spring-run Chinook and the unknown-run yearling Chinook, which is not surprising given that some of the unknown-run (spring, summer) Chinook were spring-run Chinook, but could not be positively identified as such during sampling at Rock Island Dam.

In addition to steelhead and yearling Chinook, predation rates on two other salmonid species (coho and sockeye salmon) and one additional Chinook salmon age-class (subyearling Chinook) were also estimated as part of research funded by the Grant County Public Utility District. These species and age-classes are not part of ESA-listed populations, nor were they intentionally tagged to be representative of the run passing Rock Island Dam, but sufficient sample sizes were interrogated passing Rock Island Dam (> 500 PIT-tagged fish). Estimates of tern predation rates were 1.5% for coho salmon (95% c.i. = 0.3-3.0; n = 801), 0.3% for subyearling Chinook salmon (95% c.i. = 0.1-0.6; n = 2,907), and 0.1% for sockeye salmon (95% ci = <0.1-0.3; n = 2,683). Results from these other species, plus results from yearling Chinook and steelhead, indicate that, relative to their abundance, predation rates by Goose Island Caspian terns were significantly higher on steelhead, were similar between yearling Chinook and coho salmon, and were lowest on subyearling Chinook and sockeye salmon. Differences in average smolt fork length between these various species and age-classes exist, with steelhead the largest, followed by yearling Chinook, coho, sockeye, and subyearling Chinook salmon(FPC 2012). Predation rate results coupled with average species-specific fork length data suggest that size selectivity may be a factor in the higher tern predation rates on steelhead compared to salmon (see below for additional details), a hypothesis supported by published studies conducted at other Caspian tern colonies in the region (Collis et al. 2001, Ryan et al. 2003, Hostetter et at. 2012).

Caspian tern predation rates from this and other studies indicate that the Upper Columbia River steelhead DPS is the ESA-listed salmonid population most likely to benefit from management actions to reduce avian predation in the Columbia Plateau region (Lyons et al. 2011b). Caspian terns nesting at Goose Island/Potholes have consistently had the highest predation rates on Upper Columbia River steelhead smolts released at Rock Island Dam (10.8% to 21.9%; Table 8 and Figure 53). Avian predation on Upper Columbia River steelhead in 2013 was not limited to Caspian terns from this one breeding colony, however, with predation rates at other colonies in 2013 ranging from a low of 0.1% to a high of 6.0% (Table 8). Taken together (all piscivorous waterbird colonies combined), between 31.0% and 47.3% of Upper Columbia River steelhead smolts annually released into the tailrace of Rock Island Dam during 2008 to 2013 were consumed by avian predators before reaching the ocean (Table 8).

A growing body of evidence suggests that behavioral and physical traits associated with hatchery-raised salmonids enhance susceptibility to predation (Olla and Davis 1989, Johnson and Abrahams 1991, Fritts et al. 2007, Hostetter et al. 2012). For instance, predation rates by Caspian terns nesting at Goose Island in Potholes Reservoir during 2013 were significantly higher for hatchery-reared steelhead smolts (17.2%; 95% c.i. = 14.5-20.5%; Table 6) compared to wild steelhead smolts (8.5%; 95% c.i. = 6.0-11.4%; Table 6). Similar differences in susceptibility to avian predation were observed at other colonies of fish-eating birds in 2013 (see Table 6). This trend was not universal, however, with predation rates on wild fish sometimes as large, if not larger, than those on hatchery fish (Table 6), depending on the salmonid ESU and bird colony. Intrinsic (e.g., fork length) and extrinsic (e.g., run-timing) factors may have contributed to the lower predation rates on wild steelhead smolts relative to hatchery-reared smolts by Goose Island terns in 2013.

Within populations of Pacific salmonids, higher susceptibility to avian predation has been attributed to differences in abundance, condition, size, rearing, and environmental conditions (Collis et al. 2001, Schreck et al. 2006, Kennedy et al. 2007, Evans et al. 2012, Hostetter et al. 2012). In this study, steelhead and yearling Chinook smolts out-migrating later in the season (late May through June) were generally more susceptible to predation by Caspian terns nesting at Goose Island in Potholes Reservoir (Figure 54). Conversely, Caspian tern predation rates were often lowest during the period of peak out-migration (Figure 54). Thus, the relative abundance of fish in the river may also be an important factor regulating predation rates, whereby the odds of an individual fish being consumed decrease as more fish enter the population (Hostetter et al. 2012). Similar temporal trends in steelhead susceptibility to predation by Goose Island Caspian terns were observed in 2008-2012 (BRNW 2013a). Individual smolt characteristics, such as fork length and external condition, were also associated with differences in susceptibility to avian predation. Predation rates by Caspian terns nesting at Goose Island were highest on steelhead with fork lengths of 17 - 24 cm, and lower for steelhead that were longer (> 24 cm fork length) or shorter (< 15 cm fork length; Figure 55). Wild steelhead smolts were often shorter than their hatchery-reared counterparts (Figure 55), which may have contributed to lower Caspian tern predation rates on wild steelhead smolts compared to their hatchery-reared counterparts. Due to the observational nature of this study, however, it is not known whether lower predation rates on smaller steelhead smolts were due to predator foraging strategies (i.e., selection for fish of certain lengths) or prey behavior (i.e., behavioral differences between smaller, often wild smolts and longer, often hatchery-raised smolts).

Caspian terns nesting at Goose Island in Potholes Reservoir are known to disproportionately consume steelhead in degraded condition (BRNW 2013a). External condition of out-migrating steelhead smolts has previously been used as a metric of health and linked to internal fish condition (Hostetter et al. 2011, Connon et al. 2012), steelhead survival during out-migration (Hostetter et al. 2011), susceptibility to avian predation (Hostetter et al. 2012), and adult returns (Evans et al. 2014). In 2013, steelhead exhibiting increased levels of body damage and disease symptoms suffered higher avian predation rates compared to steelhead without these external anomalies (Figure 56). Smolt condition alone did not explain all differences in avian predation rates, however, especially considering the low prevalence of steelhead with external damage (Figure 56). Similar condition-dependent analyses on yearling Chinook sampled at Rock Island Dam in 2013 were not possible because predation rates on yearling Chinook were so low (1.7%) and because few yearling Chinook were observed with moderate-to-severe external anomalies (< 5% of sampled fish).

Ultimately, the probability of an individual fish surviving the juvenile life stage is determined by a complex set of interacting factors, including individual fish characteristics and environmental conditions (Skalski 1998, Muir et al. 2001, Zabel et al. 2005, Hostetter et al. 2011). Non-lethal external examination was, however, able to identify several individual fish characteristics and environmental factors that were correlated with increased susceptibility to avian predation. Differences in avian predation rates as a function of smolt fork length, condition, and run-timing indicated that a representative sample of PIT-tagged smolts (i.e., not culled by fork length, condition, rearing-type, or run-timing) is required to accurately estimate the impact of avian predation at the level of the salmonid ESU/DPS. Such estimates are a valuable source of information for fisheries managers and salmonid population monitoring programs.

In past years, there has been a discrepancy between Goose Island Caspian tern predation rates on steelhead smolts estimated from PIT tag recoveries and estimates of steelhead consumption rates based on bioenergetics modeling. This discrepancy was especially large in 2013. In 2013, new information on the foraging behavior of Goose Island Caspian terns derived from terns fitted with GPS tracking devices (see Appendix C) was collected. These data indicated that some Goose Island Caspian terns commuted long distances to the upper Columbia River and lower Snake River, and those that did made significantly fewer foraging trips of longer duration than Caspian terns that foraged in Potholes Reservoir and other locations close to the colony. The Caspian terns that commuted to the Columbia and Snake rivers thus had fewer opportunities to transport fish back to the colony and, consequently, salmonids were under-represented in tern diet data collected at the colony. Anecdotal evidence of greater prey availability in Potholes Reservoir in 2013 suggests that terns foraging locally may have been able to make shorter duration foraging trips and more frequently provision mates and chicks than in other years, causing this bias in diet composition data based on bill load identification to be more extreme than in other years. Bioenergetics estimates of Caspian tern consumption of juvenile salmonids, based on diet data collected at the Goose Island colony, should be considered minimum estimates of smolt consumption. Predation rates on steelhead smolts that were derived from PIT tag recoveries on-colony are a less-biased estimate of the impact on the Upper Columbia River steelhead population of Caspian terns nesting at Goose Island in Potholes Reservoir.

Crescent Island Caspian terns – Following the nesting season, a total of 6,398 PIT-tagged smolts from the 2013 migration year (Chinook, coho, sockeye, and steelhead combined from all releases) were recovered on the Crescent Island Caspian tern colony (Table 2). Control tags sown on the Crescent Island tern colony (n = 200) indicated that detection efficiency ranged from 53% to 90% for tags deposited between 1 April and 31 July (Table 3 and Appendix D, Figure D1). Deposition rates for Crescent Island Caspian terns were estimated to be 71% (95% c.i. = 62-81%; Table 4 and Appendix A).

Of the available PIT-tagged fish last detected passing Lower Monumental Dam (Snake River ESUs) or Rock Island Dam (Upper Columbia River ESUs; Map 1), predation rates by Crescent Island terns were highest for Snake River steelhead (2.8%; 95% c.i. = 2.4-3.4%) and Upper Columbia River steelhead (2.8%; 95% c.i. = 2.2-3.5%; Table 7). Predation rates on salmon ESUs were < 1.0% for all populations evaluated in 2013 (Table 7). ESU-specific predation rates on steelhead DPSs in 2013 were similar to those in 2012 (Appendix B), with impacts to Snake River and Upper Columbia River steelhead populations greater than those on salmon populations from the same rivers (BRNW 2013a). Differences in Crescent Island Caspian tern predation rates on hatchery and wild steelhead were relatively small and non-significant (Table 6).

Crescent Island tern predation on smolts originating from rivers downstream of Lower Monumental Dam and Rock Island Dam (the two dams used to determine fish availability for predation rate estimates), but upstream of McNary Dam (i.e., within the foraging range of Caspian terns nesting at Crescent Island) on the middle Columbia River are not included here, but are likely smaller because only a fraction of smolts originating from these ESUs are susceptible to tern predation (i.e., large numbers of smolts from middle Columbia River ESUs enter the river downstream of McNary Dam and thus have a low susceptibility to predation by Caspian terns nesting at Crescent Island; Evans et al. 2012). It is also important to note that portions of out-migrating smolts from Snake River ESUs are captured at dams and put aboard barges for transportation downstream and release below Bonneville Dam. These transported smolts are not exposed to predation by Crescent Island terns or any other avian predators in McNary, John Day, The Dalles, and Bonneville pools. This means that the impact on each Snake River ESU from avian predation in McNary Pool is less than indicated by the predation rate on the in-river migrating portion of the ESU. Transportation rates of Snake River smolts vary considerably by year and species, with between 20% and 65% of available Snake River smolts collected for transportation during 2007-2012 (FPC 2013). An estimate of transportation rates for Snake River smolts in 2013 is not yet available, but because some proportion of these ESUs were transported in 2013 the over-all impact of Crescent Island terns on Snake River ESUs is less than those presented in Table 7. Because smolts originating from Upper Columbia River ESUs, however, are not collected for transportation above McNary Dam, the estimated predation rate on in-river migrants applies to all available fish belonging to the ESU/DPS (i.e., no correction for transportation is needed for Upper Columbia River ESUs).

In general, predation rates on juvenile salmonids by Crescent Island terns in 2013, based on all ESUs/DPSs evaluated, were similar to those in years past (Appendix B). This indicates that predation rates on salmonids by Crescent Island terns have remained relatively constant over the last few nesting seasons. Similarly, the size of the tern colony has remained relativity stable, ranging from 349 to 422 pairs since 2008 (Figure 6).

Blalock Island Caspian terns – PIT tag recovery was conducted in the Blalock Islands this year, but only 26 pairs of Caspian terns attempted to nest on the Blalock Islands in 2013 (Figure 11). Following the nesting season, just 135 PIT-tagged smolts from the 2013 migration year (Chinook, coho, sockeye, and steelhead combined from all releases) were recovered (Table 2). Due to the paucity of PIT tags recovered (Table 3) and high on-colony tern deposition rate estimates (71%; Table 4 and Appendix A), predation rates were below ≤ 0.1% for all ESUs/DPSs evaluated. It is worth noting, however, that of 135 PIT tags recovered, 105 were from steelhead smolts.

1.4.3 Coastal Washington

There was no attempt to recover salmonid PIT tags from Caspian tern colonies in coastal Washington during 2013.

1.4.4 Interior Oregon and Northeastern California

Methods: Similar to anadromous salmonids from the Columbia River basin, Warner suckers, Lost River suckers, shortnose suckers, and Klamath largescale suckers are PIT-tagged to evaluate their behavior and survival following release. With the exception of Klamath largescale suckers, all of these sucker species are ESA-listed. In 2013 we continued to evaluate the impacts of Caspian terns nesting at Corps-constructed islands located at Crump Lake, Tule Lake, and Sheepy Lake (Map 2) on survival of suckers by recovering PIT tags on these islands after the nesting season. In addition to scanning for sucker PIT tags, the Corps-constructed tern island on Malheur Lake was scanned for PIT tags, a location where redband trout from the nearby Donner and Blitzen River were PIT-tagged by the Oregon Department of Fish and Wildlife (ODFW) as part of USACE-funded study. Results from Malheur Lake redband trout study will be reported by ODFW (POC, Shaun Clements).

Results and Discussion: The Crump Lake, Tule Lake, and Sheepy Lake tern islands have been scanned for sucker PIT tags following each of the nesting seasons when breeding Caspian terns were present during 2008-2013. Only one sucker PIT tag has been recovered on a tern colony during this seven-year period, a PIT tag recovered following the 2008 nesting season on the Crump Lake tern island. This PIT tag was from a 22-cm Warner sucker that was captured and released by ODFW into Crump Lake in June 2008 (Paul Sheerer, ODFW, pers. comm.). In 2013, a PIT tag from a juvenile sucker (species unknown) was found on the Sheepy Lake tern island, but the tag was not found in an area of the island used by nesting Caspian terns, but instead along the edge of the island where American white pelicans and double-crested cormorants frequently roost.

The small number (n = 1) and percentage (< 0.1%; BNRW 2013) of Warner sucker PIT tags recovered on the Crump Lake Caspian tern colony suggests that mortality to Warner suckers from predation by Caspian terns nesting at Crump Lake island has been extremely low since the island was built during the winter of 2007-08. With the exception of the lone PIT tag found at an off-colony, mixed-species loafing area on Sheepy Lake in 2013, no sucker PIT tags have been recovered at the Tule Lake tern island or the Sheepy Lake tern island to date, which also suggests that mortality of shortnose and Lost River suckers due to predation by Caspian terns in the Upper Klamath Basin is extremely rare.

Although no PIT-tagged suckers have been found on Caspian tern colonies in the Upper Klamath Basin, juvenile suckers were occasionally observed in Caspian tern bill loads (see Section 1.3.4), indicating that Caspian terns do consume suckers, albeit rarely. Because these suckers were juveniles, however, it is not known whether they were ESA-listed suckers or non-listed suckers (e.g., Klamath largescale suckers) because juvenile suckers in this region cannot be identified to the level of species based on morphology (D. Hewitt, USGS, pers. comm.).

Finally, sucker PIT tags have been found on the breeding colonies of piscivorous colonial waterbirds other than those of Caspian terns (e.g., double-crested cormorants and American white pelicans), both in the Warner Valley and in the Upper Klamath Basin, in previous years (see BRNW 2011 for details). Results from these efforts indicate that pelicans and cormorants may pose a risk to sucker survival, especially juvenile sucker survival, although more research is needed to accurately quantify impacts and to more fully understand predator-prey interactions.

1.5. Color Banding and Band Re-sightings

In 2013, we continued our efforts to band breeding adult Caspian terns and Caspian tern chicks near fledging age at several nesting colonies as part of an on-going demographic study. The banding efforts are also part of our continuing studies of movement rates by adult terns among breeding colonies. Results presented here track the movements of banded Caspian terns among colonies, either within or between years, to better assess the consequences of various management initiatives implemented as part of the Caspian Tern Management Plan for the Columbia River Estuary.

1.5.1. Columbia River Estuary

Methods: In 2013, Caspian tern chicks near fledging age were banded with a federal numbered metal leg-band and two colored plastic leg-bands on one leg and a colored plastic leg-band engraved with a unique alphanumeric code on the other leg. This compliment of bands allows us to individually identify each banded tern from a distance, such that the banding location (colony) and banding year are known. Tern chicks that were too small to be color-banded were banded with a federal numbered metal leg-band only. Tern chicks were captured by herding flightless young into holding pens located at the periphery of the colony. Once captured, Caspian terns were immediately transferred to small crates designed to hold birds until they were banded and released. Banding operations were conducted during periods of moderate temperatures to reduce the risk of heat stress for captive terns.

Caspian terns that were color-banded in previous years (2001 – 2012) were re-sighted on the East Sand Island tern colony by researchers using binoculars and spotting scopes during 5-7 days per week throughout the 2013 breeding season. Numbers of banded Caspian terns re-sighted with a complete set of color bands, thus identifying banding location and year are presented in this report.

Multi-state analysis (Hestbeck et al. 1991, Brownie et al. 1993) in program MARK (White and Burnham 1999) was used to estimate inter-colony (or inter-region, in some cases) movement probabilities of Caspian terns banded as adults during 2005-2012. *A priori* models were constructed to evaluate effects of transitions from one colony (or region) to another and effects of year on movement probabilities. Akaike's Information Criterion (AIC) adjusted for small samples (AICc) was used to select the best model (Burnham and Anderson 2002) to estimate inter-colony movements. Based on movement probabilities between 2012 and 2013 estimated from the best model, and the numbers of Caspian terns present at a colony in 2012, net movements (the estimated number of terns that moved from one colony to another, subtracted from the number of terns that moved in the opposite direction) between 2012 and 2013 were estimated.

Results and Discussion: In 2013, 264 Caspian tern chicks near fledging age were color-banded on East Sand Island; 2 smaller tern chicks were only banded with metal leg bands.

In 2013, a total of 737 previously color-banded Caspian terns were re-sighted at the East Sand Island tern colony. Of the 737 re-sighted terns, 684 (93%) were banded at East Sand Island (246 as adults and 438 as chicks), 14 (2%) were banded at Crescent Island (3 as adults and 11 as chicks), 12 (2%) were banded at Goose Island/Potholes (8 as adults and 4 as chicks), 11 (1%) were banded as chicks at Dungeness Spit on the Olympic Peninsula, 10 (1%) were banded as chicks at the Port of Bellingham, Washington, and 5 (1%) were banded as chicks at Brooks Island or Knight Island in San Francisco Bay, California. Finally, 1 (< 1%) was banded as a chick at the Crump Lake tern island in the Warner Valley, Oregon. Re-sightings of banded Caspian terns at the East Sand Island colony indicate that some terns are moving from both inland and coastal colonies to the East Sand Island colony.

Out of 19 *a priori* models constructed in 2013, a model with transition (from one colony to another) and year effects on inter-colony movement probabilities was selected based on the smallest value of AICc. This model included an interaction term between transition and year effects, which allows movement probabilities to vary over years regardless of trends observed in other transitions. It better explained movement probabilities than a model used in 2012, which had an additive term between transition and year effects.

There was little movement (< 0.01%) of Caspian terns banded as adults from East Sand Island to Crescent Island prior to 2010. During 2010-2013, there were limited movements from East Sand Island to Crescent Island, < 1 % in most years with the exception of a 1.8% movement rate in 2012. Estimated net movement rate from East Sand Island to Crescent Island in 2013 was 41 individuals (Appendix D, Table D6). Movement probabilities of Caspian terns banded as adults from the East Sand Island colony to the Goose Island-Potholes colony ranged from <0.01% to 0.6% during 2010-2013. Estimated net movement from East Sand Island to Goose Island in 2013 was 8 individuals.

Movement rates of terns banded as adults from the East Sand Island colony to alternative colony sites on the Corps-constructed tern islands in interior Oregon and northeastern California (all sites were lumped together and considered as one region in this analysis) ranged from < 0.01% to 5.3% during 2008-2013, with the highest movement probability recorded in 2013 (Appendix D, Table D6). This 2013 movement probability translates into an estimated net movement of 659 terns from East Sand Island to the Corps-constructed islands in 2013.

The substantially higher estimated net movement of adult Caspian terns from East Sand Island to the Corps-constructed tern islands, compared to that from East Sand Island to other colonies in 2013, suggests that the on-going implementation of the Caspian Tern Management Plan for the Columbia River Estuary, which is designed to redistribute part of the East Sand Island colony to the alternative colonies in interior Oregon and northeastern California, is effective to some extent.

1.5.2. Columbia Plateau

Methods: In 2013, adult Caspian terns and Caspian tern chicks near fledging age were banded at the Crescent Island and Goose Island tern colonies. Adult Caspian terns were captured using noose mats placed around active nests. The methods for capturing chicks and banding adults and chicks were the same as those described in Section 1.5.1.

Terns that were color-banded in previous years were re-sighted during 1-2 days per week at the Crescent colony and 3-4 days per week at the Goose Island colony throughout the breeding season in 2013. The methods to estimate inter-colony movement probabilities were the same as those described in Section 1.5.1.

Results and Discussion: At the Crescent Island Caspian tern colony in 2013, 67 adult terns and 153 tern chicks near fledging age were color-banded; 16 smaller tern chicks were only banded with metal leg-bands. At the Goose Island Caspian tern colony in 2013, 43 adult terns and 117 tern chicks near fledging age were color-banded; 35 smaller tern chicks were only banded with metal leg-bands.

In 2013, a total of 258 previously color-banded Caspian terns were re-sighted at the Crescent Island colony. Of these, 221 (86%) were banded at Crescent Island (122 as adults and 99 as chicks), 27 (10%) were banded at Goose Island or Solstice Island in Potholes Reservoir (22 as adults and 5 as chicks), and 10 (4%) were banded at East Sand Island (2 as adults and 8 as chicks). In 2013, a total of 210 previously color-banded Caspian terns were re-sighted at the Goose Island colony in Potholes Reservoir. Of these, 143 (68%) were banded at Goose Island or Solstice Island (107 as adults and 36 as chicks), 49 (23%) were banded at Crescent Island (11 as adults and 38 as chicks), 14 (7%) were banded at East Sand Island (2 as adults and 12 as chicks), 2 (1%) were banded as chicks at Port of Bellingham, and 2 (1%) were banded at Brooks Island in San Francisco Bay.

Based on the best model selected to estimate inter-colony movements (see section 1.5.1), movement probabilities of Caspian terns banded as adults from Crescent Island to Goose Island ranged from < 0.01% to 10.6% during 2010-2013, with movement probability of 2.8% in 2013 (Appendix D, Table D6). Movement probabilities from Goose Island to Crescent Island ranged from < 0.01% to 16.5% during 2011-2013, with a movement probability of 9.9% in 2013. The estimate of net movement between the two colonies was 69 terns from Goose Island to Crescent Island in 2013, which may partly explain the decline in colony size at Goose Island in 2013.

Although the numbers are small, a few adult Caspian terns present at East Sand Island in the Columbia River estuary in 2012, where management actions to reduce the size of the colony are being implemented, moved to Crescent Island and Goose Island in 2013 (see section 1.5.1). Caspian tern movements from East Sand Island to colonies in the Columbia Plateau region could off-set benefits to salmonids of tern management in the estuary because per bird impacts on smolt survival are higher for terns nesting in the Columbia Plateau region compared to those nesting in the estuary, where marine forage fishes (anchovy, smelt, surfperch, etc.) tend to dominate the diet.

1.5.3. Interior Oregon and Northeastern California

Methods: The methods for capturing and banding of Caspian tern chicks at colonies on Corps-constructed islands in interior Oregon and northeastern California were the same as those described in Section 1.5.1. Caspian terns that were color-banded in previous years were re-sighted during 4-5 days per week at the Corps-constructed tern islands on Malheur Lake and Crump Lake, 1-4 days per week at Sheepy Lake and Tule Lake Sump 1B, and 1 day per week at Summer Lake Wildlife Area (East Link and Gold Dike) throughout the 2013 breeding season.

Results and Discussion: A total of 112 Caspian tern chicks near fledging age were color-banded and 51 smaller chicks were banded with metal leg-bands at only two colonies on the Corps-constructed tern islands in interior Oregon and northeastern California during 2013: Sheepy Lake and Malheur Lake. At the Sheepy Lake tern colony, 72 tern chicks were color-banded and 35 smaller chicks were banded with metal leg bands only. At the Malheur Lake tern colony, 40 tern chicks were color-banded and 16 smaller chicks were banded with metal leg bands only.

A total of 82 color-banded Caspian terns were re-sighted at the Corps-constructed tern island at Crump Lake in 2013; 35 (43%) were banded at Crump Lake (11 as adults and 24 as chicks), 16 (20%) were banded at Crescent Island (1 as an adult and 15 as chicks), 11 (13%) were banded as chicks at Goose Island, 8 (10%) were banded at East Sand Island (1 as an adult and 7 as chicks), 8 (10%) were banded at the Sheepy Lake tern island (4 as adults and 4 as chicks), 2 (2%) were banded as chicks at the Tule Lake tern island, 1 (1%) was banded as an adult at Summer Lake, and 1 (1%) was banded as a chick at Dungeness Spit.

A total of 5 color-banded Caspian terns were re-sighted at the Corps-constructed island in East Link impoundment at Summer Lake Wildlife Area during 2013. Of the five color-banded terns that were re-sighted at the East Link tern island, 3 were banded as chicks at Crescent Island, and 2 were banded at Goose Island (1 as an adult and 1 as a chick).

A total of 127 color-banded Caspian terns were re-sighted at the colony on the Corps-constructed island at Sheepy Lake in Lower Klamath NWR during 2013; 52 (41%) were banded at East Sand Island (3 as adults and 49 as chicks), 21 (17%) were banded as chicks at Crescent Island, 20 (16%) were banded at Goose Island (8 as adults and 12 as chicks), 12 (9%) were banded at the Sheepy Lake tern island (7 as adults and 5 as chicks), 12 (9%) were banded at Crump Lake tern island (2 as adults and 10 as chicks), 5 (4%) were banded as adults at the Tule Lake tern island, 2 (2%) were banded as chicks at Brooks Island or Knight Island in San Francisco Bay, 1 (1%) was banded as an adult at Summer Lake, 1 (1%) was banded as a chick at the Port of Bellingham, and 1 (1%) was banded as a chick at Dungeness Spit.

A total of 161 color-banded Caspian terns were re-sighted at the colony on the Corps-constructed island at Tule Lake Sump 1B in Tule Lake NWR during 2013; 58 (36%) were banded at East Sand Island (2 as adults and 56 as chicks), 23 (14%) were banded as chicks at Crescent Island, 29 (18%) were banded at Goose Island or Solstice Island in Potholes Reservoir (9 as adults and 20 as chicks), 16 (10%) were banded at Sheepy Lake tern island (5 as adults and 11 as chicks), 14 (9%) were banded at Crump Lake (1 as an adult and 13 as chicks), 7 (4%) were banded as adults at Tule Lake tern island, 8 (5%) were banded as chicks at Port of Bellingham, 4 (2%) were banded as chicks at Brooks Island or Knight Island in San Francisco Bay, and 2 (1%) were banded as chicks in the Copper River Delta, Alaska.

A total of 318 color-banded Caspian terns were re-sighted at the Corps-constructed tern island in Malheur Lake during 2013; 131 (41%) were banded at Crescent Island (27 as adults and 104 as chicks), 87 (27%) were banded at Goose Island or Solstice Island in Potholes Reservoir (34 as adults, 53 as chicks), 49 (15%) were banded at East Sand Island (2 as adults and 47 as chicks), 38 (12%) were banded at Crump Lake (10 as adults and 28 as chicks), 8 (3%) were banded at Sheepy Lake tern island (1 as an adult and 7 as chicks), 3 (1%) were banded as adults at Tule Lake tern island, 1 (<1%) was banded as a chick at the Port of Bellingham, and 1 (<1%) was banded as a chick at Brooks Island in San Francisco Bay.

Re-sightings of banded Caspian terns at the recently established colonies on the Corps-constructed islands in interior Oregon and northeastern California Lake revealed high inter-colony connectivity, both among coastal and interior breeding colonies. Caspian terns banded at East Sand Island were re-sighted at four different Corps-constructed islands that were built by the Corps as alternative tern nesting habitat in interior Oregon and northeastern California as part of the Caspian Tern Management Plan for the Columbia River estuary; all of these recently built tern islands are more than 400 km from East Sand Island. Movements of banded Caspian terns among the Corps-constructed alternative nesting islands in interior Oregon and northeastern California were also documented.

SECTION 2: DOUBLE-CRESTED CORMORANTS

2.1. Nesting Distribution and Colony Size

2.1.1. Columbia River Estuary

Methods: High-resolution, vertical aerial photography of the double-crested cormorant colony on East Sand Island was taken during the late incubation period to estimate the peak size of the colony. Three independent counts of the number of attended nests visible in aerial photography were used to estimate the total number of breeding pairs; standard errors from these counts were used to estimate a confidence interval for this estimate. Beginning in 2008, we expanded the use of aerial photography to estimate colony size across the entire breeding season. High resolution aerial photography of the cormorant colony was taken approximately every 2 weeks from early May to early September in 2013. Aerial photography that included the entire East Sand Island cormorant colony was taken nine times during the 2013 nesting season (including the photography taken late in incubation to estimate peak colony size). We developed a custom application in ArcGIS to count nests or individual birds on all of the aerial photography of the East Sand Island cormorant colony, as well as to count aerial photography of breeding colonies of other piscivorous colonial waterbirds (e.g., terns, gulls, and pelicans).

Boat-based and aerial surveys of double-crested cormorants nesting on 12 navigational markers near Miller Sands Spit and Fitzpatrick Island (river km 38 and 53, respectively) in the Columbia River estuary were conducted once a month from mid-April through early July in 2013. Because nesting chronology varied among the different channel marker groups, the number of cormorant breeding pairs at each marker group was estimated using the greatest number of attended nests observed on each group of markers throughout the season. Any well-maintained nest structure attended by a cormorant adult and/or chick was considered active. To minimize impacts to nesting cormorants (i.e., chicks jumping from nests into the water when disturbed), we did not climb navigational markers and check nests to estimate productivity.

Four boat-based surveys of nesting cormorants on the Astoria-Megler Bridge in the Columbia River estuary were conducted from mid-April to early July 2013. Our vantage point from a boat enabled us to count the number of attended cormorant nests on the underside of the bridge; however, visual confirmation of eggs or very small chicks in nests was not possible. Any well-maintained nest structure that was attended by an adult cormorant was considered active, along with any nests that contained visible chicks.

Periodic boat-, land-, and air-based surveys were also conducted to monitor the sites where double-crested cormorants previously nested on Rice Island and on Miller Sands Spit. During these surveys, researchers looked for indications of cormorant nesting activity.

Results and Discussion: In 1989 fewer than 100 pairs of double-crested cormorants nested on East Sand Island. Growth in the size of the breeding colony since 1989 has resulted in the East Sand Island colony becoming the largest known colony of double-crested cormorants in western North America (Adkins and Roby 2010). We estimate that 14,916 breeding pairs (95% c.i. = 14,545 – 15,287 breeding pairs) attempted to nest at the East Sand Island colony in 2013, a significant increase in colony size compared to the previous year (12,301 breeding pairs; Figure 57). The size of the East Sand Island double-crested cormorant colony grew rapidly from 1997 to 2007, nearly tripling in size (Figure 57 and Appendix D, Table D7). In 2008, however, the colony experienced an unexpected decline (20%) before rebounding to the peak colony size observed in 2013 (Figure 57). The growth of the East Sand Island colony appears to be exceptional among colonies of double-crested cormorants along the coast of the Pacific Northwest, where most colonies are stable or declining. The available data suggest that much of the early growth of the East Sand Island colony was a result of immigration from colonies outside the Columbia River estuary. More data are needed to assess the extent to which factors limiting the size and reproductive success of colonies throughout the Pacific Northwest are influencing the size of the double-crested cormorant colony on East Sand Island.

Prior to 1999, double-crested cormorants on East Sand Island nested exclusively on the boulder riprap and driftwood at the southwest corner of the island. After 1999 they began nesting on the ground in satellite colonies in the adjacent low-lying habitat. Based on the apparent habitat preferences of nesting double-crested cormorants, there is currently ample unoccupied habitat on East Sand Island, which could support further expansion of the colony in the future (Map 5). Despite availability of habitat to support continued colony expansion, bald eagle disturbance and predation, plus the associated nest predation by glaucous-winged/western gulls (*Larus glaucescens/occidentalis*), may currently be limiting the size of the double-crested cormorant colony on East Sand Island. In 2013, nesting density for cormorants on East Sand Island was 1.20 nests/m^2, within the range in nesting densities observed at the colony over the past 17 years (0.73 – 1.28; Appendix D, Table D7). It is likely that suitable nesting habitat for double-crested cormorants on East Sand Island is not limiting, with further expansion of the colony possible.

In 2013, we surveyed for double-crested cormorants nesting on 12 channel markers located in the upper Columbia River estuary, eight near Miller Sands Spit and four near Fitzpatrick Island. A maximum of 330 pairs of double-crested cormorants nested on 11 of these channel markers, eight near Miller Sands Spit and three near Fitzpatrick Island; this total number of active nests is higher compared to the 2012 count (248 breeding pairs on 10 channel markers). Counts of attended cormorant nests at both groups of channel markers peaked in late June.

Double-crested cormorants continued nesting near the pelagic cormorant (*P*. *pelagicus*) colony on the Astoria-Megler Bridge in 2013. In addition, thousands of double-crested cormorants were observed roosting on the bridge at various times throughout the breeding season, possibly associated with the nest dissuasion activities on East Sand Island in 2013 (see Section 2.6). During four boat-based censuses from 25 April to 19 July, a maximum of 231 active double-crested cormorant nests were counted on the Astoria-Megler Bridge; this count was a large increase from 2012, when 139 double-crested cormorant nests were counted on the bridge.

2.1.2. Columbia Plateau

Methods: Counts of attended cormorant nest structures were used to estimate the size of the double-crested cormorant colony on Foundation Island in the mid-Columbia River during 2013 (Map 1). To enhance the accuracy of nest counts, we constructed an observation platform in the water, approximately 90 m off the eastern shore of the island. Nest counts and observations of nest contents were conducted periodically to estimate colony size and determine if double-crested cormorants were successful in rearing young at the colony in 2013. In addition, for Foundation Island, aerial images were collected in series to include virtually the entire cormorant colony to investigate feasibility of using aerial images to estimate breeding colony size. The aerial photographer collected a linear image series during each of 6-8 flight passes of the colony. Images were reviewed later and the most complete and clear image series was loaded into a GIS and attended nest structures were counted for an estimate of colony size.

In 2013, we conducted three aerial surveys of the Columbia Plateau region (25-26 April, 15-16 May, and 2-3 July) looking for new breeding colonies of double-crested cormorants. Additionally, periodic land- and boat-based surveys were conducted throughout the breeding season to verify nesting by cormorants at sites identified during aerial surveys. At each site we counted attended cormorant nests to obtain an estimate of the number of breeding pairs at each colony. Where aerial photography was taken near the peak in breeding and images provided a more complete view of a breeding colony (compared to boat- or ground-based observation points), photo counts of attended nests were used to estimate colony size. Although it is possible that some small colonies (i.e., < 20 breeding pairs) may have been missed during these surveys, we are confident that all breeding colonies of consequence within the study area, that did not fail early in the nesting season, were identified.

Results and Discussion: In 2013, the double-crested cormorant colony on Foundation Island consisted of a minimum of 386 breeding pairs based on counts from the observation platform on 3 May. Foundation Island continues to be the largest cormorant breeding colony on the mid-Columbia River. All nesting at this cormorant colony occurs in trees. During 2003-2006 the Foundation Island cormorant colony gradually grew from about 250 breeding pairs to about 360 breeding pairs, before leveling off and then declining to about 310 breeding pairs during 2009-2011 (Figure 58). In 2012 and 2013, the size of the Foundation Island cormorant colony reached its largest to date, at around 390 breeding pairs (Figure 58). Data on weekly colony attendance in 2013 was not collected at the Foundation Island cormorant colony.

Aerial images collected on 26 April corresponded closest in time to the date (3 May) when the platform-based data indicated a peak in breeding and were used to assess correspondence of aerial-based methods and traditional colony size estimation for double-crested cormorants nesting on Foundation Island. Two counts of the total number of attended nests visible on the 26 April images were 374 and 380 for an estimate of 377 breeding pairs. Correspondence of this estimate with the platform-based count of 363 attended nests recorded on 27 April and the peak count of 386 nests was relatively good. Thus, for Foundation Island, it appears that aerial images of the cormorant colony collected at or near the peak of breeding may provide a reasonable index of colony size.

While attended double-crested cormorant nest counts from aerial images of Foundation Island corresponded well with platform-based counts and aerial images of ground nesting birds provide clear views and relatively complete counts for some species (e.g., Harper Island double-crested cormorants), several factors may constrain accurate estimation of colony size from imagery. First, timing of photography to occur during or near the breeding peak is needed. For colonies that experience early breeding failure, or delayed nest initiation, counts from photography scheduled for the average or anticipated date of peak breeding may underestimate colony size. Additionally, vegetation may obscure some tree or ground nesting birds and vegetation growth or decline may lead to variable effects on nest visibility within and across breeding seasons. Finally, the size and complexity of some colonies (e.g., North Potholes Reserve) make collection of photography and estimation of colony size from photos very difficult.

As was the case in 2012, no nesting by double-crested cormorants was observed on the mid-Columbia River at either Crescent Island or Miller Rocks in 2013. In 2011, cormorants attempted to nest at both of these sites (1 and 2 breeding pairs, respectively) and subsequently failed in their nesting attempts. The cause(s) of nest failure at these incipient colonies was not determined.

The largest double-crested cormorant colony in the Columbia Plateau region is at Potholes Reservoir in the North Potholes Reserve, where ca. 800 breeding pairs nested in 2013 (Figure 59). This colony was in decline from its peak colony size in 2006 (ca. 1,150 breeding pairs) to 2009 (ca. 810 breeding pairs) and has since been gradually increasing in every year since, until it declined to the lowest level ever observed in 2013 (Figure 59). As with the Foundation Island colony, cormorants at the North Potholes colony nest in trees, and at North Potholes the trees are flooded for much of the nesting season. Although this colony is the largest of its kind in the region, there is little evidence that these birds commute to the Columbia River to forage on juvenile salmonids, based on the scarcity of salmonid PIT tags beneath the colony.

Based on our counts of attended cormorant nests at the Okanogan cormorant colony at the mouth of the Okanogan River, we estimate that there was a minimum of 42 breeding pairs at this colony in 2013, similar to the count the previous year (40 breeding pairs).

We estimated that 174 breeding pairs of double-crested cormorants nested at the colony on Harper Island in Sprague Lake during 2013, more than in 2012 (146 breeding pairs). We first observed cormorants nesting on Harper Island in 2008, when an estimated 38 breeding pairs nested on the island. Double-crested cormorants were also recorded nesting on Harper Island in the early 1990s (M. Monda, Washington Department of Fish and Wildlife, pers. comm.). Harper Island is also home to a large California and ring-billed gull colony and a small Caspian tern colony.

Aerial surveys of the lower and middle Columbia River from Bonneville Dam to Rock Island Dam, and the lower Snake River from its mouth to the confluence with the

Clearwater River, revealed no other breeding colonies of double-crested cormorants in 2013.

There was of total of four active double-crested cormorant colonies in the Columbia Plateau region during 2013, where a total of approximately 1,406 breeding pairs nested (Figure 60). The breeding population of double-crested cormorants in the Columbia Plateau region had been increasing in every year over the past 4 years until 2013, when it declined to just above the 8-year average (Figure 61).

2.1.3. Coastal Washington

Methods: In 2013, we surveyed for double-crested cormorant nests on 12 channel markers in Grays Harbor, WA during three aerial survey flights from late April to late June. No boat-based surveys of nesting cormorants in Grays Harbor were conducted during 2013.

Results and Discussion: We counted a maximum of 143 double-crested cormorant nests on eight different channel markers during the aerial survey of Grays Harbor in late May, which is slightly more than the count of nesting cormorants in Grays Harbor in 2012 (130 double-crested cormorant nests on nine channel markers).

2.1.4. Interior Oregon and Northeastern California

Methods: In 2013, we conducted two aerial surveys of interior Oregon and northeastern California (05 June, 10 July; Map 4) looking for breeding colonies of double-crested cormorants. Additionally, periodic land- and boat-based surveys were conducted throughout the breeding season to verify nesting by cormorants at sites identified in aerial surveys.

Results and Discussion: Based on aerial, land, and boat-based surveys in 2013, double-crested cormorants were confirmed nesting at nine different locations: Upper Klamath NWR (ca. 1,071 individuals counted at six nesting sites), Sheepy Lake in Lower Klamath NWR (82 individuals), Sump 1B in the Tule Lake NWR (261 individuals), Clear Lake NWR (ca. 327 individuals counted at two nesting sites), River’s End Ranch near Valley Falls, OR (4 breeding pairs), Gold Dike impoundment at Summer Lake Wildlife Area (143 individuals), Malheur NWR (ca. 268 individuals at two colony sites, Sodhouse Farm and Singhus Ranch), Carmine Ditch near Burns, OR (2 breeding pairs), and Crane Prairie Reservoir (23 breeding pairs). Colony counts were derived from photography taken during the aerial survey on 5 June 2013.

2.2. Nesting Success

2.2.1. Columbia River Estuary

Methods: Elevated blinds located in the East Sand Island cormorant colony were used to observe nesting cormorants in 2013 (Map 5). These blinds were accessed via above-ground tunnels to prevent disturbance to nesting cormorants and gulls, as well as roosting California brown pelicans.

In 2013, nesting attempts by 266 pairs of double-crested cormorants in eight separate plots were monitored for productivity. Observations of nest contents were recorded each week from mid-April through July to determine nesting chronology and monitor nesting success. Productivity was measured as the number of nestlings in each monitored nest at 28 days post-hatching. Cormorant chicks older than 28 days are capable of leaving their nests. Productivity was averaged for each plot and the standard error of those averages was used to calculate a confidence interval for the overall productivity estimate.

Monitoring of nesting cormorants on channel markers in the upper estuary and on the Astoria-Megler Bridge was conducted periodically (1 – 2 times per month) from a boat.

Results and Discussion: We estimated that 35,217 fledglings (95% c.i. = 29,682 – 40,752 fledglings) were produced at the East Sand Island cormorant colony in 2013. This corresponds to an average productivity of 2.36 young raised per breeding pair (95% c.i. = 2.00 – 2.72 fledglings/breeding pair), the highest productivity observed at the colony over the past 4 years (Figure 62). A proximate factor limiting double-crested cormorant nesting success on East Sand Island in recent years has been predation and associated disturbance by bald eagles from late May to late June, a limiting factor that was not as pronounced in 2013 compared to the previous three years.

Confirmation of eggs in cormorant nests on channel markers and on the Astoria-Megler Bridge was not possible from our vantage on the water, but the first double-crested cormorant chicks were observed on 12 June at the channel markers and on 19 July at the Astoria-Megler Bridge colonies during the 2013 nesting season. Due to poor vantage and infrequent visits, we were not able to estimate nesting success for double-crested cormorants that nested on the upper estuary channel markers or on the Astoria-Megler Bridge.

2.2.2. Columbia Plateau

Although estimates of nesting success are not available for the double-crested cormorant colonies in the Columbia Plateau region, these four colonies (i.e., Foundation Island, North Potholes, Harper Island, and Okanogan River) were successful in raising some young to fledging age in 2013.

2.2.3. Coastal Washington

It is unknown whether double-crested cormorants nesting on channel markers in Grays Harbor were successful in raising young to fledging age in 2013. No aerial surveys were conducted over Grays Harbor during the chick-rearing or fledging periods.

2.2.4. Interior Oregon and Northeastern California

Methods: Breeding colonies of double-crested cormorants in interior Oregon and northeastern California were photographed during aerial surveys or visited late in the breeding season to determine if they were successful in raising young.

Results and Discussion: Double-crested cormorants nesting at Clear Lake NWR, Malheur Lake, Upper Klamath NWR, Tule Lake NWR, Summer Lake Wildlife Area, and Sheepy Lake in Lower Klamath NWR were likely successful in raising some young to fledging age in 2013. Nesting success at the other double-crested cormorant colonies in interior Oregon and northeastern California (i.e., River's End Ranch, OR; Carmine Ditch, OR; and Crane Prairie Reservoir, OR) during 2013 was not determined.

2.3. Diet Composition and Salmonid Consumption

2.3.1. Columbia River Estuary

Methods: Lethal sampling techniques were necessary to assess the diet composition of double-crested cormorants nesting on East Sand Island. The best method to obtain a random sample of the diet is to collect adult birds commuting toward the colony from foraging areas throughout the breeding season. The target sample size for collections was 5-15 samples of adult foregut (stomach and esophagus) contents per week. This sampling effort was selected to adequately capture seasonal changes in diet while minimizing the impact of lethal sampling to the colony as a whole. Immediately after collection, each cormorant's abdominal cavity was opened, the foregut removed, and the contents of the foregut emptied into a whirl-pak. Each foregut sample was weighed, labeled, and stored frozen for later sorting and analysis in the laboratory.

Analysis in the laboratory of semi-digested diet samples was conducted at Oregon State University. Samples were partially thawed, removed from whirl-paks, re-weighed, and separated into identifiable and unidentifiable fish soft tissue. Fish in cormorant foregut samples were identified to genus and species, whenever possible. Intact salmonids in foregut samples were identified as Chinook salmon, sockeye salmon, coho salmon, steelhead, or unknown based on genetics analyses[2]. Unidentifiable fish soft tissue samples were artificially digested (work that is ongoing) according to the methods of Petersen et al. (1990, 1991). Once digested, diagnostic bones (i.e., otoliths, cleithra, dentaries, and pharyngeal arches) were removed from the sample and identified to species using a dissecting microscope (Hansel et al. 1988). Unidentified fish soft tissue samples that did not contain diagnostic bones and samples comprised of bones only (i.e., no soft tissue) were excluded from diet composition analysis. Taxonomic composition of double-crested cormorant diets was expressed as % of identifiable prey biomass. The prey composition of cormorant diets was calculated for each 2-week period throughout the nesting season. The diet composition of double-crested cormorants over the entire breeding season was based on the average of these 2-week percentages.

Estimates of annual smolt consumption by double-crested cormorants nesting at the East Sand Island colony were calculated using a bioenergetics modeling approach (Roby et al. 2003, Lyons 2010). We used a Monte Carlo simulation procedure to estimate 95% confidence intervals for estimates of smolt consumption by double-crested cormorants.

Results and Discussion: Based on identifiable fish tissue in foregut samples (92% of the collected biomass of stomach contents), juvenile salmonids comprised 10.7% of the diet (by biomass) of double-crested cormorants nesting at East Sand Island in 2013 (n = 134 adult foregut samples or a total of 25,281 g of identifiable fish tissue). The average proportion of juvenile salmonids in the diet of double-crested cormorants nesting on East Sand Island in 2013 was roughly half the percentage during the previous year (20.0%), and was slightly below the 14-year average (11.8%; Figure 63).

The diet of double-crested cormorants, which forage by pursuit-diving throughout the water column, at the East Sand Island colony is more diverse (Figure 64) than that of Caspian terns nesting on the same island (Figure 36). On average, northern anchovy was the single most prevalent prey type for double-crested cormorants nesting at East Sand Island in 2013, followed by various marine, estuarine, and freshwater taxa (Figure 64 and Figure 65). In 2013, the prey category “other” consisted of more than eight different taxa, all less than 5% of the diet, with the exception of stickleback, which was 8% of the diet by biomass. In recent years, the proportion of the diet consisting of salmonids has increased from a low of 2% of identifiable biomass in 2005, to a high of 20% in 2012. Over this time period, anchovy has been the single most prevalent prey type in cormorant diets, but annual values have varied from 23-34% of the diet by biomass. The

[2] Genetic analyses were conducted by NOAA Fisheries (POC: David Kuligowski) at the Manchester Field Station genetics laboratory. Species identifications were carried out by amplifying (PCR) the mitochondrial DNA fragment COIII/ND3 as outlined in Purcell et al. (2004). Following species identification, samples were genotyped using species-specific standardized sets of microsatellite DNA markers (Seeb et al. 2007; Blankenship et al. 2011). Stock origins of individual salmon and steelhead were estimated using standard genetic assignment methods (Van Doornik et al. 2007).

increasing proportion of salmonids in cormorant diets during 2010-2012 had generally been associated with small declines in many of the less common prey types (e.g., surf perch, cyprinids, others), which moderately rebounded in prevalence during 2013. The seasonal peak in the proportion of salmonids in the diet of double-crested cormorants nesting on East Sand Island during 2012 was in late May (20% by biomass; Figure 66).

Genetic stock identification of salmonid samples collected from double-crested cormorant stomachs during 2011-13 indicated that cormorants consumed smolts from many of the uniquely identifiable stocks from across the basin. For Chinook salmon in the cormorant diet during April and May, the most common genetic stock of origin was Snake River spring Chinook salmon (16 of 34 or 47% of Chinook samples from this period; Figure 67 and Appendix D, Table D8). During June and July, the most prevalent Chinook salmon stock (19 of 31 or 61%) originated from the lower Columbia River (Spring Creek Group fall run, West Cascades Tributary fall run, and the introduced Rogue River fall run stocks). Identified steelhead trout originated from five stocks, with steelhead from the Snake River consisting of just over half of the identified samples (32 of 63 samples or 51%; Figure 68 and Appendix D, Table D4). A majority of coho salmon in the cormorant diet were of Columbia River origin; however, coho salmon identified as originating from the Oregon coast stock made up 20% of samples (22 of 60), and two samples was identified as originating from the Washington coast stock (Figure 68 and Appendix D, Table D5).

Our estimate of total smolt consumption by double-crested cormorants nesting on East Sand Island in 2013 was 16.3 million smolts (95% c.i. = 11.4 – 21.1 million). The best estimate of total smolt consumption was lower than in 2011 and 2012, but the 95% confidence intervals substantially overlapped (Figure 69). An estimated 11.4 million sub-yearling Chinook smolts (95% c.i. = 6.9 – 15.9 million; 70% of total smolt consumption) were consumed by double-crested cormorants from the East Sand Island colony in 2013, very similar to the 2012 estimate (10.8 million). Estimated consumption of spring migrating smolts (coho, yearling Chinook, and sockeye salmon along with steelhead) was 4.8 million smolts (95% c.i. = 3.8 – 5.8 million). This represented a significant decrease in consumption of spring migrants compared to 2012 (8.1 million smolts [95% c.i. = 6.2 – 9.9 million smolts]). The consumption of these species/types in 2013 were 2.7 million coho salmon smolts (95% c.i. = 2.0 – 3.4 million; 17% of total smolt consumption), 1.0 million steelhead smolts (95% c.i. = 0.8 – 1.3 million; 6%), 0.9 million yearling Chinook salmon smolts (95% c.i. = 0.6 – 1.1 million; 5%), and 0.2 million sockeye salmon smolts (95% c.i. = 0.05 – 0.3 million; 1%; Figure 70). In general, annual smolt consumption by double-crested cormorants nesting on East Sand Island remained high in 2013 (Figure 69), due to continued high consumption of sub-yearling Chinook salmon smolts and the increase in size of the colony. Consumption of spring migrating smolts has been variable in recent years, however. In 2013, consumption of these smolts returned to levels seen during 2010-2011, after a higher level of consumption in 2012 (Figure 70). During 2010-2013, estimates of total smolt consumption by East Sand Island cormorants have been significantly higher than that of Caspian terns nesting on

East Sand Island (Figure 41); however, in 2013 and some other recent years, consumption of spring migrants has been more similar for both avian predators (4.8 million spring migrant smolts consumed by cormorants and 3.5 million spring migrant smolts consumed by Caspian terns [95% c.i. = 3.0 – 4.0 million smolts]). Cormorant colony size has been relatively stable since 2006; changes in colony size have not explained recent variability in cormorant smolt consumption. The primary factor driving varying estimates of smolt consumption by double-crested cormorants during 2010-2012 has been a variable proportion of smolts in the cormorant diet in these years (Figure 63). Cormorant consumption of spring migrant smolts was significantly lower in 2013 (an estimated 41% lower) compared to 2012, despite a 21% increase in peak colony size.

2.3.2. Columbia Plateau

No diet composition data were collected for double-crested cormorants in the Columbia Plateau region 2013.

2.3.3. Coastal Washington

No diet composition data were collected for double-crested cormorants nesting along the Washington coast in 2013.

2.3.4. Interior Oregon and Northeastern California

Although no diet composition data were collected for double-crested cormorants nesting outside the Columbia River basin, PIT tags from ESA-listed suckers were recovered on mixed-species colonies of piscivorous waterbirds (which included double-crested cormorants) in interior Oregon and northeastern California; see Section 3.3.4 for those results.

2.4. Predation Rates Based on Salmonid Smolt PIT Tag Recoveries

2.4.1. Columbia River Estuary

Methods: The methods for calculating predation rates on juvenile salmonids based on PIT tag recoveries at the East Sand Island double-crested cormorant colony are the same as those described for Caspian terns in Section 1.4.

Results and Discussion: Following the nesting season, 11,029 PIT-tagged smolts (Chinook, coho, sockeye, and steelhead combined from all releases) from the 2013 migration year were recovered on the East Sand Island double-crested cormorant colony (Table 2). Control tags to measure detection efficiency were sown during two discrete time periods (pre- and post-season) and indicated that detection efficiency ranged from 62% to 72% for tags deposited between 1 March and 31 August (Table 3 and Appendix D, Figure D2). Equal numbers of control tags were sown pre-season (n = 200) and post-season (n = 200) on the colony (Table 3). Unfortunately, 100 of the 200 pre-season tags sown on-colony were incidentally sown in a segment of the colony were birds did not actual nest in 2013. To account for this known bias, these 100 PIT tags were excluded from detection efficiency calculations and an average, pre-season detection efficiency from years past was used to estimate pre-season detection efficiency in 2013. All 200 post-season tags were sown in areas occupied by nesting cormorants during the 2013 nesting season and were used to generate detection efficiency estimates. Deposition rates for East Sand Island cormorants were estimated to be 60% (95% c.i. = 47 - 73%; Table 4 and Appendix A).

Predation rates on PIT-tagged smolts last detected passing Bonneville Dam on the Columbia River or Sullivan Dam on the Willamette River (Map 1) ranged from 0.7% (95% c.i. = 0.3 - 1.4%) on Upper Willamette River spring Chinook to 2.9% (95% ci = 2.4-3.7%) on Snake River spring Chinook (Table 5). Predation on steelhead DPSs, a species that is normally highly susceptible to predation by double-crested cormorants nesting on East Sand Island, ranged from 1.6% (95% c.i. = 0.7 - 2.6%) on Middle Columbia River steelhead to 2.7% (95% c.i. = 1.9 - 3.7%) on Upper Columbia River steelhead (Table 5). For those salmonid populations with adequate sample sizes (> 500 PIT-tagged fish, per rear-type), there was no significant difference in predation rates between hatchery and wild fish belonging to the same ESUs/DPSs (Table 5), a finding consistent with results from previous years (Evans et al. 2011a, Lyons et al. 2012).

Predation rate estimates by East Sand Island cormorants on salmon and steelhead ESUs/DPSs in 2013 were lower than those reported in recent years, especially predation rates on steelhead populations. For example, Lyons et al. (2012) reported an average annual predation rate of 9.8% (range = 3.9 - 18.5%) on Snake River steelhead during 2007-2011, nearly five times the value observed in 2013 (Table 5). Furthermore, predation rates on steelhead populations in 2013 were similar to those on salmon populations (Table 5), a finding that is inconsistent with data collected in previous years when steelhead populations were depredated at a higher rate than salmon populations. These results are especially surprising given that the double-crested cormorant colony on East Sand Island in 2013 was the largest recorded since monitoring begin in 1998 (Figure 57). ESU-specific predation rate estimates from 2013 and estimates of total smolt consumption in 2013 (see Section 2.3.1) both indicate that impacts on salmonid survival were lower in 2013 compared to 2012, and that colony size was not a good predictor of impacts on salmonids in 2013. Research to identify factors other than colony size that influence the impact of cormorant predation on salmonids in the estuary – such as marine forage fish availability, river discharge, ocean conditions, and others – are currently being investigated with funding by the USACE as part of management plans to reduce cormorant predation in the estuary. Results of this investigation may help managers correctly attribute a decline in cormorant predation rates to a particular management action, or to correctly interpret why a management action to reduce colony size may have caused little in the way of a reduction in smolt mortality in any given year.

Data on the impacts of predation by double-crested cormorants and Caspian terns nesting on East Sand Island (see Section 1.4.1) on survival of PIT-tagged salmonid populations from the Lower Columbia River (LCR) are not available in 2013. Estimates of predation rates for LCR DPSs/ESUs are not available because a representative sample of PIT-tagged fish by location (geographic boundary, including releases below Bonneville Dam), by origin (hatchery, wild), and by outmigration timing were lacking. An analysis of predation rates on LCR DPSs/ESUs conducted by Lyons et al. (2012), which attempted to account for these data gaps as well as possible, indicated that 26% and 28% of available LCR Chinook and coho salmon, respectively, were depredated by double-crested cormorants nesting on East Sand Island each year during 2007-2010. Although Lyons et al. (2012) concluded that more research was needed to understand the impact of double-crested cormorant predation on LCR DPSs/ESUs, the limited data available suggest that LCR Chinook and coho salmon may be more susceptible to predation by double-crested cormorants nesting on East Sand Island compared to DPSs/ESUs originating further upriver (Sebring et al. 2013).

2.4.2 Columbia Plateau

There were no attempts to recover smolt PIT tags from double-crested cormorant colonies in the Columbia Plateau region in 2013. Predation rates by double-crested cormorants nesting on Foundation Island, WA; in North Potholes Reserve, WA; and on Harper Island in Sprague Lake, WA can be found in previous BRNW Annual Reports (2008-2013).

2.5. Color banding

Methods: In 2013, we continued to band double-crested cormorant adults and chicks at the colony on East Sand Island in the Columbia River estuary with a federal numbered metal leg-band on one leg and a field-readable plastic leg-band engraved with a unique alphanumeric code on the other leg. This was the sixth year of a prospective long-term effort to collect information on the demography (survival and recruitment), inter-colony movements, and dispersal patterns of double-crested cormorants from the East Sand Island colony using re-sightings of banded individuals.

Double-crested cormorants were captured for banding using several methods in 2013. Prior to active hazing of pre-nesting adult cormorants on a portion of the East Sand Island colony during mid-April (see Section 2.6), night-time capture using large landing nets and spotlights was employed to obtain adult cormorants for banding and tagging from the dissuasion area (Map 5). Adult cormorants were also captured at night (May-June) from above-ground access tunnels that concealed our presence on the portion of the cormorant colony where nesting occurred. Breeding adult cormorants were captured during late incubation. Cormorant chicks near fledging age (> 28 days post-hatch) were also captured at night from the above-ground tunnels for banding purposes. Finally, some cormorant chicks were captured for banding when they were near fledging age during a daytime round-up near the perimeter of the cormorant colony. Once captured, cormorants were transported to an adjacent processing area, banded, and released.

Double-crested cormorants banded with color bands during 2008-2013 were re-sighted from multiple observation blinds on the cormorant colony at East Sand Island. Re-sighting efforts were a minimum of 12 hours of observation per week from all blinds combined during most of the breeding season.

Results and Discussion: A total of 201 adult double-crested cormorants and 420 double-crested cormorant chicks were captured, banded, and released at the East Sand Island colony in 2013. Of the 201 adult cormorants that were banded in 2013, 108 (54%) were captured in the nest dissuasion area during mid-April (83 of which were also fitted with satellite tags; see Section 2.6). An additional 93 adult cormorants were captured from tunnels located on the active portion of the cormorant colony, between the privacy fences (see Map 5). Of the 420 cormorant chicks banded in 2013, 407 (97%) were captured from the above-ground tunnels and 13 were captured during the daytime round-up.

We have banded a total of 1,961 double-crested cormorants (816 adults during 2010-2013 and 1,145 chicks during 2008-2013) with field-readable color bands at East Sand Island. To date, only a small proportion (an estimated 2-3%) of all breeding adult double-crested cormorants nesting at East Sand Island have been banded. About 69% (n = 560) of double-crested cormorants banded as adults at East Sand Island during 2010-2013 were observed at least once on East Sand Island in 2013. Approximately 1% of young raised at the East Sand Island colony in 2013 were banded before fledging. A total of 102 double-crested cormorants that were banded as chicks in previous years were observed at East Sand Island in 2013. Some cormorants banded as chicks in 2012 were observed returning to the East Sand Island colony in 2013. The age of double-crested cormorants that were banded as chicks in previous years and confirmed breeding in 2013 ranged from 2- to 5-years-old.

While a significant effort was expended to re-sight banded cormorants on East Sand Island during the 2013 breeding season, re-sighting efforts at other regional colonies remained infrequent and opportunistic. Encounters with banded cormorants away from East Sand Island are limited, but have increased in recent years, with most observations occurring between breeding seasons and reported by the public. From January 2013 to date, we have received reports of 36 banded double-crested cormorants observed alive or dead from eight different regions outside the Columbia River estuary: the Salish Sea/Puget Sound region (banded cormorants re-sighted alive: n = 3; banded cormorants recovered dead: n = 5); western British Columbia (re-sighted: n = 9; recovered: n = 2);

outer Washington coast (recovered: n = 1); interior Washington (recovered: n = 1), Oregon coast (re-sighted: n =2; recovered: n = 1); Willamette Valley (re-sighted: n = 2; recovered: n = 3); California coast (re-sighted: n = 4; recovered: n = 1); and interior California (recovered: n = 2). Although there were no banded cormorants re-sighted in the Columbia Plateau region during 2013, one cormorant banded at the East Sand Island colony was re-sighted near Richland, WA during the post-breeding season 2012.

Continued banding and re-sighting efforts will allow us to measure movement rates of double-crested cormorants from East Sand Island to other colonies and roosting sites, and to measure demographic parameters for estimating population trajectories. That is critical information for both predicting and assessing the outcome of various prospective management strategies for double-crested cormorants nesting on East Sand Island.

2.6. Management Feasibility Study

Methods: In 2011, 2012, and 2013, we tested the feasibility of dissuading double-crested cormorants from nesting on a portion of their former colony on East Sand Island using privacy fences, nest destruction, and targeted human hazing (BRNW 2012, 2013a). The privacy fences provided a visual barrier that concealed the activities of the hazers to nesting cormorants located in the non-dissuasion areas of the colony (see Map 5). In 2010, the year before the large scale dissuasion experiments started, there was a total of 16 acres of habitat available to nesting cormorants on the western end of East Sand Island (BRNW 2013b). In 2011, one privacy fence was erected that bisected the colony on its eastern end and double-crested cormorants were dissuaded from nesting in ca. 6%, or about 1 acre, of the available nesting habitat in 2010, leaving them 15 acres of available nesting habitat west of the privacy fence. In 2012, the privacy fence was moved further west and cormorants were dissuaded from nesting in ca. 31%, or about 5 acres, of the available habitat in 2010, leaving them with 11 acres of available nesting habitat west of the privacy fence. In 2013, two privacy fences were erected in the same locations as the fences erected in 2011 and 2012 (2.4 m high by 25 m long and 65 m long, respectively; see Map 5), and hazing was used to prevent cormorants from nesting east of the eastern fence and west of the western fence. In 2013, double-crested cormorants were dissuaded from nesting in ca. 75%, or about 12 acres, of the available nesting habitat in 2010, leaving them with 4 acres of habitat for nesting between the two privacy fences (see Map 5).

Not all of the available cormorant nesting habitat on the western end of East Sand Island (16, 15, 11, and 4 acres in 2010, 2011, 2012, and 2013, respectively) is utilized by nesting cormorants (BRNW 2013b). Based on the amount of habitat occupied by nesting cormorants on East Sand Island over the past 16 years, generally less than 3 acres (BRNW, unpubl. data), the dissuasion experiment in 2013 still provided ample habitat to accommodate all of the cormorants that previously nested on East Sand Island.

In addition to the two privacy fences, two camps, six observation blinds, and an above-ground tunnel system were constructed to provide researchers access to the area without disturbing nesting cormorants outside of the targeted dissuasion areas. The camps concealed all routine non-hazing researcher activity from cormorants within the dissuasion area, as well as those cormorants nesting between the privacy fences, and the blinds provided an elevated vantage point for observations on both sides of the privacy fences.

Cormorants were first observed in the dissuasion area on 11 April and hazing efforts began on 13 April. The dissuasion area was scanned every half hour from dawn to dusk during each day. During each scan, researchers counted cormorants in the dissuasion area and recorded breeding behaviors (i.e., courtship displays, nest building, copulations). Researchers flushed cormorants from the dissuasion area when (1) double-crested cormorants exhibited breeding behaviors, (2) aggregations of 100 or more loafing cormorants were observed in the dissuasion area, or (3) cormorants were present in the dissuasion area prior to civil twilight in the evening; the latter was in order to prevent overnight roosting in the dissuasion area. If no hazing occurred for two hours, the frequency of scans was reduced to every hour. To minimize disturbance to other birds in the dissuasion area (i.e., roosting brown pelicans and nesting glaucous-winged/western gulls), researchers only remained visible on the cormorant colony until cormorants had dispersed and then immediately returned to camp. In addition, researchers adopted best management practices (e.g., standardized travel paths that avoided established gull nesting areas and avoided direct eye contact with gulls and brown pelicans) to further minimize research disturbance to non-target species. Following dissuasion activities, researchers remained in the blind to conduct post-dissuasion observations to determine the effectiveness of hazing activities, enumerate any disturbance to brown pelicans, and assess disturbance to cormorants nesting between the privacy fences. At least one researcher monitored the dissuasion area from 11 April until 30 June, when all daily cormorant dissuasion activities ceased for the season.

Under permit, a limited number of double-crested cormorant eggs (not to exceed 250) could be removed from nests in the dissuasion area, if some cormorants laid eggs despite efforts to prevent egg-laying. Egg take would be used to enhance the prospects of successful nest dissuasion on a portion of the East Sand Island cormorant colony and was not used as a means of limiting or reducing nesting success at the colony.

To evaluate where displaced double-crested cormorants might prospect for alternative nest sites if they left the East Sand Island colony, we captured and marked 109 adult double-crested cormorants in the dissuasion area during 12 - 16 April, shortly after their arrival in that part of the colony. All captured double-crested cormorants were banded with a federal numbered metal leg-band on one leg and a field-readable plastic leg-band engraved with a unique alphanumeric code on the other leg. Of the 109 banded double-crested cormorants, 83 were fitted with satellite transmitters and 26 only received bands.

Battery-powered satellite tags weighing ca. 50 g were attached as backpacks using a harness made of Teflon ribbon (Dunstan 1972), modified as described by King et al. (2000). The satellite tags were programmed to collect nighttime roost locations every night or every other night through July, then once a week from August through March, before switching back to the more frequent cycle in April. The tags transmitted nighttime roost location data to the ARGOS satellite network and data were later retrieved from the website of CLS America, Inc.

Results and Discussion: As was the case the previous two years, the nest dissuasion experiment conducted in 2013 was effective at preventing cormorants from nesting in the targeted dissuasion areas. Double-crested cormorants were hazed in the western dissuasion area a total of 57 days between 13 April and 30 June, while hazing occurred on just 11 days in the eastern dissuasion area between 26 April and 13 June. As many as 2,700 cormorants were observed in the dissuasion areas prior to the commencement of hazing activities on 13 April. An average of four (range = 0 - 21) hazing incursions were conducted in the dissuasion areas each day, with the number dependent upon the return rate and subsequent behavior of cormorants in each dissuasion area. While cormorants continued to prospect in dissuasion areas throughout the study period, no cormorant eggs were known to have been laid in either of the two dissuasion areas.

Impacts to non-target birds/species associated with the nest dissuasion experiment in 2013 were believed to be minimal. Dissuasion activities caused little or no disturbance to cormorants nesting between the privacy fences. For example, double-crested and Brandt's cormorants established nests within 1 meter of the eastern privacy fence and successfully raised young at those nests. Furthermore, more than 16,000 breeding pairs of cormorants (double-crested and Brandt's) nested between the two privacy fences in 2013, each having above-average nesting success. California brown pelicans roosted in and adjacent to the dissuasion area throughout the active hazing period, with up to 3,500 brown pelicans observed roosting in these areas at times. Brown pelicans were disturbed during nine cormorant hazing events; a maximum of 500 individual brown pelicans were flushed during one of these hazing events. The last of the cormorant hazing events occurred on 30 June, several weeks prior to the influx of large numbers of brown pelicans to East Sand Island that were seen in previous years. Lower brown pelican use of East Sand Island in 2013 may have been due to regional factors influencing pelican distribution (e.g., forage fish availability in other parts of the California Current ecosystem). Despite our hazing activities, several thousand glaucous-winged/western gulls still nested and raised young in the cormorant dissuasion areas. Since glaucous-winged/western gulls nest across the entire island (BRNW 2013b), nesting gulls that may have been displaced as a result of our hazing activities likely re-nested elsewhere on the island.

Immediately following deployment of satellite tags on double-crested cormorants captured in the dissuasion area, some of the tagged cormorants left the Columbia River estuary (defined as from the river mouth to Puget Island [Rkm 74.5]). About 96% (80 of 83) of the cormorants satellite-tagged in 2013 dispersed from the East Sand Island colony after tagging, and of these about 96% (73 of 76; four tags ceased transmitting) eventually returned to East Sand Island and attempted to nest there. During this initial dispersal period (16 April – 15 May), tagged cormorants were detected at colonies and roost sites away from East Sand Island in the following locations: (1) in the Columbia River estuary (n = 71); (2) on the lower Columbia River below Hood River (n = 26); (3) on the outer Washington coast (including Willapa Bay and Grays Harbor; n = 18); (4) in the Salish Sea (n = 1); and (5) on the outer coast of Vancouver Island (n = 1). During the same period, tagged cormorants visited active cormorant breeding colonies in the Columbia River estuary (Astoria-Megler Bridge, channel markers), lower Columbia River (Troutdale transmission towers), and coastal Washington (Grays Harbor channel markers).

Tagged cormorants dispersing from East Sand Island during the nesting season (April through September) were detected at colonies and roost sites in the following locations: (1) in the Columbia River estuary (n = 76); (2) on the lower Columbia River below Hood River (n = 27); (3) on the outer Washington coast (including Willapa Bay and Grays Harbor; n = 54); (4) in the Salish Sea (n = 16); and (5) on the outer coast of Vancouver Island (n = 3). During the nesting season, tagged cormorants visited active cormorant breeding colonies in the Columbia River estuary (Astoria-Megler Bridge, channel markers); on the lower Columbia River (Troutdale transmission towers); in coastal Washington (Grays Harbor channel markers, Willoughby Rock); and in the Salish Sea (Bird Rocks, Snohomish River mouth). No confirmed detections of tagged cormorants came from inland sites east of Hood River, and there were no detections on the Oregon Coast (Map 6).

Summary: Privacy fences, nest destruction, and targeted human hazing has proven to be a feasible method of preventing double-crested cormorants from nesting in pre-defined areas of the East Sand Island cormorant colony without adversely impacting non-targeted birds/species. Nest dissuasion experiments conducted during 2011-2013 were successful in progressively limiting the available nesting habitat on the western end of East Sand Island from a total of 16 acres in 2010 down to just 4 acres in 2013. Despite this, the available nesting habitat during the nest dissuasion experiments was not reduced to the point where nesting habitat became limiting for cormorants on the western end of East Sand Island. The vast majority of double-crested cormorants hazed in 2013 were able to successfully relocate to nest in the non-dissuasion area between the two privacy fences. If these methods were used to induce cormorants from East Sand Island to nest elsewhere, the habitat would likely need to be reduced further, such that the available nesting habitat on East Sand Island became limiting (< 2.5 acres; BRNW, unpubl. data).

The general pattern of short dispersal trips and high return rates to East Sand Island, suggests that cormorants may display high colony site fidelity if resource managers decide to permanently reduce available nesting habitat in the future. High colony site fidelity may be a result of prolonged nesting history at the site (many individual cormorants having nested at East Sand Island their entire lives), social facilitation from this very large colony, nesting habitat not being limiting on East Sand Island, or the lack of suitable nesting opportunities elsewhere. To induce cormorants to permanently emigrate from the Columbia River estuary, it may be necessary to further restrict nesting habitat on East Sand Island and prevent greater use of alternative nesting sites within the estuary (e.g., the Astoria-Megler Bridge).

See BRNW (2013b, 2013c) for more information on the nest dissuasion feasibility studies on East Sand Island during 2008-2013. Data on the dispersal of radio- and satellite-tagged double-crested cormorants from the colony on East Sand Island to other locations, dispersal that was associated with the feasibility studies of nest dissuasion techniques on East Sand Island, can be found in a geospatial database currently in development under contract with the USACE - Portland District.

SECTION 3: OTHER PISCIVOROUS COLONIAL WATERBIRDS

3.1. Distribution

3.1.1. Columbia River Estuary

Methods: In 2013, land-based, boat-based, and aerial surveys were conducted throughout the breeding season to locate piscivorous waterbird colonies in the Columbia River estuary. When possible, counts from the ground or from photography were used to estimate the number of adult birds or the number of active nests on colonies in 2013. Counts of the number of adults on colonies on East Sand Island are available for glaucous-winged/western gulls, ring-billed gulls, and Brandt's cormorants in 2013 (based on one count of adults on-colony in aerial photography; see Section 1.2.1 for a description of methods), and are presented here. Peak numbers of California brown pelicans using East Sand Island as a nighttime roost in 2013 were determined by periodic boat-based surveys conducted in the evening from mid-May through mid-September.

Results and Discussion: A total of seven nesting colonies of piscivorous waterbirds other than Caspian terns and double-crested cormorants (i.e., glaucous-winged/western gulls, ring-billed gulls, Brandt's cormorants, pelagic cormorants, and American white pelicans) were identified at four different locations in the Columbia River estuary: East Sand Island, Rice Island, Miller Sands Spit, and the Astoria-Megler Bridge. In addition, East Sand Island was once again the location of a large post-breeding, nighttime roost for California brown pelicans.

Gulls − Based on surveys conducted in 2013, glaucous-winged/western gulls nested at colonies on East Sand Island and Rice Island, while ring-billed gulls nested just on East Sand Island (Map 1). Glaucous-winged/western gulls were not confirmed breeding on Miller Sands Spit, where they previously nested in 2012. Based on one count of aerial photography taken of East Sand Island on 8 June, we estimate that ca. 4,580 glaucous-winged/western gulls and ca. 2,680 ring-billed gulls were on their respective colonies.

California Brown Pelicans − East Sand Island is the largest known post-breeding nighttime roost site for California brown pelicans, and the only known night roost for this species in the Columbia River estuary (Wright 2005). In 2013, the first California brown pelicans were observed roosting on East Sand Island on 22 April, a month later than was observed in 2012. The weekly count of brown pelicans roosting on East Sand Island peaked in late August at about 3,850 roosting birds, significantly less than the peak counts in 2011 (ca. 14,225 individuals) and 2012 (ca. 10,600 individuals). As was the case in 2009, 2010, and 2012, we observed breeding behavior by brown pelicans roosting on East Sand Island (i.e., courtship displays, nest-building, attempted copulations) in 2013. In 2013, we also observed five eggs in three nests, the first confirmed egg-laying by brown pelicans on East Sand Island. All three nests failed before chicks hatched, however. The cause of nest failure was not determined, but bald eagle activity was the most common source of non-researcher related disturbance to brown pelicans roosting on East Sand Island in 2013.

American White Pelicans – The first nesting by American white pelicans in the Columbia River estuary was recorded at Miller Sands Spit in the upper Columbia River estuary during 2010; roughly 100 adults were counted on-colony on 1 July 2010. American white pelicans have nested on Miller Sands Spit in every year since 2010, with a minimum of 104 adults attending nests on the island in 2013. While estimates of nesting success are not available, American white pelicans were successful in raising some young at the Miller Sands Spit colony in each year during 2010-2012 (these data were unavailable in 2013).

Brandt's and Pelagic Cormorants − A small colony of Brandt's cormorants consisting of 44 breeding pairs became established on East Sand Island within the double-crested cormorant colony in 2006. The numbers of Brandt's cormorants breeding on East Sand Island have since increased steadily, and in 2012 about 1,684 pairs of Brandt's cormorants nested on East Sand Island (Figure 71). In 2013, the size of the Brandt's cormorant colony on East Sand Island was about 1,523 breeding pairs, the first decline in colony size since the colony was first established in 2006 (Figure 71). This Brandt's cormorant colony is now one of the largest of its kind in Oregon and Washington, and the only site in the Columbia River estuary where Brandt's cormorants are known to nest. Before 2006, a small breeding colony of Brandt's cormorants existed on the pile dike at the western end of East Sand Island, but the site was abandoned after a storm damaged the pile dike during the winter of 2005-2006. Brandt's cormorants were first documented to nest on this pile dike in 1997, when a few pairs were found nesting there (Couch and Lance 2004). While estimates of nesting success are not available, Brandt's cormorants have been successful in raising young at the East Sand Island colony in recent years, including 2013.

At least 72 breeding pairs of pelagic cormorants nested on the Astoria–Megler Bridge in 2013, fewer than were observed the previous year (106 breeding pairs). This is the only site in the Columbia River estuary where pelagic cormorants are known to nest. Pelagic cormorants have been observed nesting on the underside of the southern portion of the Astoria-Megler Bridge since we began surveying the structure in 1999.

3.1.2. Columbia Plateau

Methods: In 2012, we conducted aerial surveys in the Columbia Plateau region looking for colonies of piscivorous waterbird species other than Caspian terns and double-crested cormorants (i.e., California gulls, ring-billed gulls, and American white pelicans). Additionally, periodic land- and boat-based surveys were conducted throughout the breeding season to verify nesting by piscivorous waterbirds at colony sites identified during aerial surveys. For colonies of special interest, high-resolution, vertical aerial photography was taken during the late incubation period and three independent counts of individual birds were conducted using an in-house GIS workstation to estimate colony size in 2013.

Results and Discussion: A total of nine gull colonies and one American white pelican colony were identified in the Columbia Plateau region during 2013.

Gulls – In 2013, California and/or ring-billed gulls were confirmed nesting at colonies on five different islands in the Columbia River between The Dalles Dam and Rock Island Dam: Miller Rocks (river km 333); "Anvil Island" and Straight Six Island in the Blalock Islands group (river km 445); Crescent Island (river km 510), and Island 20 (river km 545; Map 1). In addition, California and/or ring-billed gulls were confirmed nesting at colonies on four different islands located in the Columbia Plateau region off the mid-Columbia River: Goose Island in Potholes Reservoir, Harper Island in Sprague Lake, and two different islands near the southern end of Banks Lake (i.e., Twinning Island and Goose Island).

Estimates of gull colony size (i.e., adults on colony) on islands in the Columbia Plateau region during 2013 were obtained for Island 20 (ca. 8,980 California gulls and ca. 5,060 ring-billed gulls), Goose Island/Potholes (ca. 3,000 California gulls and ca. 9,790 ring-billed gulls), Crescent Island (ca. 5,550 California gulls and ca. 150 ring-billed gulls), Miller Rocks (ca. 4,760 California gulls and ca. 50 ring-billed gulls), Anvil Island/Blalock Islands (ca. 3,980 California gulls and ca. 1,710 ring-billed gulls), and Straight Six Island/Blalock Islands (ca. 100 California gulls and ca. 1,110 ring-billed gulls; Figure 72). Three other gull colonies (Harper Island in Sprague Lake and Twinning and Goose islands in Banks Lake) were active in 2013, but not counted. While estimates of nesting success are not available for gulls nesting in the Columbia Plateau region during 2013, all but one colony was successful in fledging at least some young (the Goose Island colony in Banks Lake completely failed in 2013 for unknown causes).

American White Pelicans − We conducted boat-based counts of American white pelicans at the colony on Badger Island in the mid-Columbia River each week during the 2013 nesting season (Map 1). Badger Island is the site of the only known nesting colony of American white pelicans in the State of Washington, and the species is listed as endangered by the State. Consequently, the island is closed to both the public and researchers in order to avoid human disturbance to nesting pelicans that might cause pelicans to abandon the breeding colony. High-resolution, vertical aerial photography was taken of the colony during the incubation period in order to estimate colony size in 2013. Complete counts of active pelican nests on Badger Island are not possible from the water or from the air because most nests are located in the interior of the island and many are concealed under thick, brushy vegetation. However, most pelicans present on the island were visible in the aerial photography. We did not correct counts of adult pelicans from aerial photography to estimate the number of breeding pairs (as we do for Caspian terns), but instead used numbers of adult pelicans from the aerial photography as an index to the number of breeding pairs utilizing Badger Island. Because it was only possible to obtain index counts of adults and juvenile pelicans at the Badger Island colony, it was not possible to estimate nesting success (average number of young raised per breeding pair).

A mean of 2,077 adult American white pelicans were counted in the aerial photography taken in 2013, similar to the colony count the previous year (2,083 adults on-colony; Figure 73). American white pelicans first nested on Badger Island in 1997 (ca. 20 breeding pairs); prior to 1997 white pelicans nested on nearby Crescent Island for several years (Figure 73). The American white pelican colony on Badger Island experienced substantial growth from 1997 to 2011, increasing by more than two orders of magnitude during that period, before leveling off in 2012 (Figure 73). Available nesting habitat on Badger Island does not appear to be a factor limiting the size of the white pelican colony.

Our boat-based counts resulted in a maximum count of ca. 50 juvenile white pelicans on 19 July 2013. For comparison, our annual maximum counts of juvenile pelicans during boat-based surveys of the Badger Island colony have ranged from 56 to 329 during the period 2002-2012.

<u>3.1.3. Coastal Washington</u>

Comprehensive surveys of nesting gulls, Brandt's cormorants, or pelagic cormorants were not conducted along the coast of Washington in 2013.

3.1.4. Interior Oregon and Northeastern California

Methods: In 2013, we conducted two aerial surveys (05 June and 10 July) in interior Oregon and northeastern California (Map 4) looking for breeding colonies of piscivorous waterbird species in addition to Caspian terns and double-crested cormorants (i.e., ring-billed gulls, California gulls, and American white pelicans). We also conducted periodic land- and boat-based surveys throughout the breeding season to verify nesting by these piscivorous colonial waterbird species at sites that were identified during aerial surveys.

Results and Discussion: Based on aerial, land-based, and boat-based surveys in 2013, gulls were confirmed to be nesting at 10 different locations and American white pelicans were confirmed to be nesting at five different locations in interior Oregon and northeastern California.

Gulls – In 2013, ring-billed and California gulls nested at six Corps-constructed islands that had been built as alternative nesting habitat for Caspian terns: the Crump Lake tern island in the Warner Valley, OR (> 1,000 breeding pairs of California and ring-billed gulls); the Sheepy Lake tern island in Lower Klamath NWR, CA (> 1,000 breeding pairs of California and ring-billed gulls); the Tule Lake tern island in Sump 1B at Tule Lake NWR, CA (2 breeding pairs of ring-billed gulls); the East Link tern island in Summer Lake Wildlife Area, OR (ca. 500 breeding pairs of California and ring-billed gulls); the Gold Dike tern island in Summer Lake Wildlife Area, OR (< 80 breeding pairs of ring-billed gulls), and the New Tern Island at Malheur NWR, OR (ca. 20 breeding pairs of California and ring-billed gulls). In addition, gull breeding colonies were noted at five additional sites during aerial surveys: Clear Lake NWR, CA (> 500 breeding pairs); Big Sage Reservoir, CA (ca. 200 breeding pairs); Meiss Lake, CA; Singhus Ranch in Malheur Lake, OR (> 300 breeding pairs); and “Old Tern Island” in Malheur Lake (> 500 pairs). Counts of gulls on the Tule Lake Sump 1B tern island in Tule Lake NWR, CA usually numbered between 50 and 100 individuals, but only two gull nests were confirmed on the island. Gulls were successful in rearing at least some young on at least six of the 11 colonies where nesting was confirmed (Crump Lake tern island, Sheepy Lake tern island, East Link tern island, Malheur Lake New Tern Island, Malheur Lake Old Tern Island, and Singhus Ranch), but were unsuccessful in rearing young at three of the 11 gull colonies (Tule Lake tern island, Gold Dike tern island, and Meiss Lake). Nesting failure at Meiss Lake was primarily attributable to dropping water levels at the two colony sites after nest initiation, which left the islands land-bridged and susceptible to terrestrial predators. Nest failure at the Tule Lake tern island can be attributed to predation by a raccoon (*Procyon lotor*) and nocturnal disturbances by great horned owls (*Bubo virginianus*). The cause of the gull nesting failure at the Gold Dike tern island was not conclusively determined, but was also likely due to disturbance by avian predators. Nesting success at the other two gull colonies in interior Oregon and northeastern California (i.e., Big Sage Reservoir and Clear Lake NWR) was not confirmed.

American White Pelicans – In 2013, American white pelicans nested at Upper Klamath NWR (< 150 breeding pairs), Clear Lake NWR (< 800 breeding pairs), and Singhus Ranch at the north side of Malheur Lake (ca. 800 breeding pairs). American white pelicans were successful in fledging some young at all three of these colonies in 2013. The island in Pelican Lake in the Warner Valley was land bridged during the 2013 breeding season and was not used by American white pelicans as a nesting site. Sheepy Lake in Lower Klamath NWR was also not used by American white pelicans as a breeding site in 2013.

3.2. Diet Composition

3.2.1. Columbia River Estuary

Gulls − We have not collected diet composition data for gulls nesting in the Columbia River estuary for over a decade. Our previous research indicated that glaucous-winged/western gulls nesting in the Columbia River estuary consumed primarily fish (Collis et al. 2002a). In general, gulls nesting on Rice Island (river km 34) ate mostly riverine fishes, whereas gulls nesting on East Sand Island (river km 8) ate primarily marine fishes. In 1997 and 1998, juvenile salmonids comprised 11% of the diet (by mass) of glaucous-winged/western gulls nesting on Rice Island/Miller Sands Spit and 4% of the diet of glaucous-winged/western gulls nesting on East Sand Island. At least some of the juvenile salmonids found in stomach samples of gulls from Rice Island/Miller Sands Spit had been kleptoparasitized (i.e., stolen) from Caspian terns, which nested at the nearby colony on Rice Island throughout the 1990s (Collis et al. 2002a). In 2012, kleptoparasitism rates (proportion of fish delivered by Caspian terns to the East Sand Island colony that were subsequently stolen by gulls) for salmonid smolts averaged 8%; steelhead smolts were kleptoparasitized by gulls at a higher rate (27%) than were salmon smolts (7%). These data indicate that gulls nesting in close proximity to Caspian terns on East Sand Island have a small impact on survival of juvenile salmonids by reducing the number of salmonid smolts successfully delivered to the tern colony.

California Brown Pelicans − Brown pelicans feed primarily on schooling marine forage fishes and, near their breeding grounds in southern California, the diet of brown pelicans consists almost entirely of anchovies (Engraulidae) and sardines (Clupeidae; Tyler et al. 1993). There is an abundance of these and other schooling marine forage fishes near East Sand Island during the summer (Emmett et al. 2006), and presumably these forage fish species comprise the majority of the diet of brown pelicans that roost on East Sand Island.

Brandt's and Pelagic Cormorants − We have not collected diet data from Brandt's or pelagic cormorants nesting in the Columbia River estuary as part of this study. Based on a study conducted in 2000, the frequency of occurrence of juvenile salmonids in the diet of Brandt's cormorants nesting in the Columbia River estuary was estimated at 7.4% (Couch and Lance 2004). Very little is known about the diet of pelagic cormorants along the Oregon coast (Hodder 2003), but the species is believed to forage primarily on marine and estuarine fishes. Due to small colony sizes and the previously-documented diet preferences of Brandt's and pelagic cormorants, the impacts of these two cormorant species on survival of juvenile salmonids from the Columbia River basin are expected to be negligible. Smolt PIT tag recoveries on the Brandt's cormorant colony on East Sand Island in 2013 support this conclusion (see Section 3.3.1).

3.2.2. Columbia Plateau

Gulls − Diet composition data analysis (from prey-items found in foregut samples; see Collis et al. 2002a) from gulls nesting on islands in the Columbia River above Bonneville Dam have not been done in over a decade. Our previous research indicated that there were small amounts of fish in general, and salmonids in particular, in the diets of California and ring-billed gulls nesting at colonies on islands in the mid- and upper Columbia River during the late 1990s (Collis et al. 2002a). The only Columbia River gull colonies where juvenile salmonids were found in diet samples were the California gull colonies on Little Memaloose Island (15% of total biomass from stomach contents; this colony is no longer active) and Miller Rocks (3% of total biomass from stomach contents; Collis et al. 2002a). Gulls from these two colonies were known to prey on juvenile salmonids at the nearby The Dalles and John Day dams (J. Snelling, OSU, pers. comm.; Zorich et al. 2010, 2011, 2012). Gulls from other up-river colonies may occasionally prey on juvenile salmonids when available in shallow pools or near hydroelectric dams (Ruggerone 1986, Jones et al. 1996), but our results from the late 1990s suggested that, at the level of the breeding colony, juvenile salmonids were a minor component of the diet. Despite this, gull colonies in the Columbia Plateau region can be large (several thousand breeding pairs; Figure 72) and impacts to survival of juvenile salmonids may, in some cases, be comparable to those of nearby Caspian tern and double-crested cormorant colonies, which are much smaller. For example, pilot studies of PIT-tag recovery on gull colonies conducted in 2012 (BRNW 2013a) suggest that predation rates on salmonids by gulls nesting at certain colonies in the Columbia Plateau region may be comparable to those of Caspian terns and double-crested cormorants nesting at colonies in the same region.

California gulls that nest at the periphery of the Caspian tern colony on Crescent Island may have a negative effect on survival of juvenile salmonids because some individuals kleptoparasitize (i.e., steal) juvenile salmonids from terns as they return to the colony to feed their mates and young. On an average foraging trip, however, breeding adult terns catch several fish and, of these fish, the majority are consumed by the adult away from the colony in order to meet the adult's energy requirements. A minority of the fish captured by a breeding adult tern are brought back to the colony to feed its mate (pre-chick rearing) or young. Only these fish are subject to kleptoparasitism by gulls. In 2012, kleptoparasitism rates on salmonid smolts delivered by Caspian terns to the Crescent Island colony averaged 19%. As was observed at East Sand Island, kleptoparasitism rates were higher on steelhead smolts (55%) than on salmon smolts (14%), suggesting that gulls prefer, or find it easier, to steal larger fish. These rates are useful in comparing gull kleptoparasitism rates among tern colonies and evaluating the relative susceptibility of different species of smolts to gull kleptoparasitism, but they are not representative of the proportion of all salmonid smolts caught by Caspian terns that were subsequently stolen by gulls (i.e., the vast majority of fish captured by terns are consumed by terns and therefore not subject to gull kleptoparasitism). Therefore, empirical data on the cumulative impacts on smolt survival associated with gull kleptoparasitism are not readily available. Given that (1) California gulls nesting at Crescent Island significantly out-number Caspian terns nesting there, and (2) gulls kleptoparasitize only a small proportion of the smolts captured by adult Caspian terns nesting at the colony (most smolts captured by terns are immediately consumed by the tern and thus not available for gulls to steal), it is unlikely that smolts kleptoparasitized by gulls fulfill more than a very small fraction of the food and energy requirements of the California gulls nesting on Crescent Island.

American White Pelicans − We did not collect data on diet composition of American white pelicans nesting on Badger Island because of the conservation status of this species in Washington. Based on smolt PIT tag detections on the Badger Island pelican colony, white pelicans do not appear to be a substantial source of mortality for smolts out-migrating in the mid-Columbia River; however, data on the rates at which PIT tags ingested by white pelicans are subsequently deposited on the Badger Island breeding colony are currently lacking (see Section 3.3.2). Regardless, the Badger Island white pelican colony may continue to grow, and an increasing number of non-breeding white pelicans have been noted along the mid-Columbia and lower Snake rivers, where they are often observed foraging below mainstem hydroelectric dams (Tiller et al. 2003, authors' unpublished data). In addition, significant numbers of white pelicans have been observed at several sites in the Yakima River basin (A. Stephenson, Yakima Klickitat Fisheries Project, pers. comm.) and elsewhere on the mid-Columbia and Snake rivers (see Section 5), pelicans that were presumably foraging on out-migrating juvenile salmonids. The impact of breeding and non-breeding American white pelicans on survival of juvenile salmonids from the upper Columbia River and Snake River basins is therefore not well understood.

3.2.3. Coastal Washington

Diet data were not collected at other piscivorous waterbird colonies along the Washington coast (see Section 3.2.1 for a general description of the diet of other piscivorous waterbirds nesting at estuary/coastal colonies).

3.2.4. Interior Oregon and Northeastern California

Diet data were not collected at other piscivorous waterbird colonies in interior Oregon and northeastern California.

3.3. Predation Rates Based on Salmonid Smolt PIT tag Recoveries

3.3.1. Columbia River Estuary

Methods: The methods for calculating predation rates on juvenile salmonids based on PIT tag recoveries at the Brandt's cormorant colony on East Sand Island are the same as those described in Section 1.4. We applied estimates of smolt PIT tag deposition rates from double-crested cormorants nesting on East Sand Island to estimate predation rates on smolts by Brandt's cormorants nesting on East Sand Island, as there are currently no estimates of on-colony PIT tag deposition rates by Brandt's cormorants. Although there are similarities in the foraging behavior, nesting behavior, and general life history between Brandt's cormorants and double-crested cormorants, it is possible that deposition rates differ between the two species.

Results and Discussion: Following the 2013 nesting season, 477 smolt PIT tags (Chinook, coho, sockeye, and steelhead combined from all releases) from the 2013 migration year were recovered on the Brandt's cormorant (BRAC) colony at East Sand Island (Table 2). The BRAC colony and the double-crested cormorant (DCCO) colony on East Sand Island were intermixed in 2013; thus, detection efficiency was measured for both colonies using the same groups of sown control tags (Table 3 and Section 2.4). Due to the degree of intermixing of the two cormorant species, recoveries of smolt PIT tags could not be definitively separated into tags deposited by BRAC versus those deposited by DCCO due to the close proximity of nests of the two species. This was much more of a concern for BRAC than DCCO because several small pockets or satellite colonies of BRAC were located on the island, pockets that were surround by larger, contiguous areas of nesting DCCO. As a result of this intermixing, estimates of smolt predation rates associated with the BRAC colony in 2013 may over-estimate (inflate) the impacts of predation by BRAC on the survival of salmonid smolts because an unknown number of the PIT tags recovered from BRAC nesting areas were deposited by DCCO.

Of the PIT-tagged juvenile salmonids last detected passing Bonneville or Sullivan dams (Map 1), predation rates by BRAC cormorants were $\leq$ 0.2% per ESU/DPS in 2013 (Table 5). Predation rates were so small that differences between species and ESUs were not discernible, or likely biologically meaningful. Again, deposition of some PIT tags by DCCO likely inflated estimates of predation rates for BRAC, so actual impacts of BRAC predation are likely less than those reported in Table 5.

Despite increases in the size of Brandt's cormorant colony on East Sand Island (Figure 71), predation rates on smolts by BRAC remained among the lowest of all the piscivorous waterbird colonies evaluated since PIT tag studies on BRAC were initiated in 2008 (BRNW 2013a). Results provide over-whelming evidence that BRAC consumed far fewer salmonid smolts per capita than do DCCO or Caspian terns nesting on East Sand Island (BRNW 2013a). Several factors likely account for this difference. First, BRAC are considered a pelagic seabird that usually forages for prey in the marine environment, where non-salmonid prey types (e.g., anchovy, herring, smelt, and others) are common. Consequently, salmonids comprise a smaller proportion of the diet of BRAC compared to that of Caspian terns and double-crested cormorants. Second, the nesting chronology of BRAC differs from that of Caspian terns and DCCO in the estuary, with colony attendance peaking in late June, compared to mid-May for Caspian terns and early June for DCCO. This difference in nesting chronology may be important because by late June the peak of the salmonid run has passed, especially for large groups of PIT-tagged steelhead and yearling Chinook salmon (FPC 2013). Finally, relative to DCCO, BRAC are smaller (by body mass), and have lower daily food requirements.

3.3.2 Columbia Plateau

Methods: The methods for calculating predation rates on juvenile salmonids based on PIT tag recoveries at California and ring-billed gull colonies on Miller Rocks, Blalock Islands (Anvil Island), Crescent Island, and Island 20 are the same as those described in Section 1.4. Data on PIT tag deposition rates were only available for California gulls nesting at these colonies in 2013 (see Appendix A). Deposition rates were assumed to be the same for both gull species (California gulls and ring-billed gulls; Table 4). Similar to potential differences in deposition rates between DCCO and BRAC (see Section 3.3.1), it is possible that differences in PIT tag deposition rates exist between California gulls and ring-billed gulls. In general, however, gull colonies evaluated in 2013 were numerically dominated by California gulls, so potential differences in deposition rates would have little, if any, impact on estimates of predation rates at these colonies.

There were no attempts to recover smolt PIT tags from the America white pelican colony on Badger Island in the mid-Columbia River in 2013. Minimum predation rate impacts from white pelicans nesting on Badger Island can be found in previous BRNW Annual Reports (2009-2013). Impacts from past annual reports are minimum estimates because no data are available to evaluate on-colony PIT tag deposition rates by American white pelicans.

Results and Discussion:

Miller Rocks Gulls – Following the 2013 nesting season, a total of 2,449 PIT-tagged smolts (Chinook, coho, sockeye, and steelhead, combined from all releases) from the 2013 migration year were recovered on the Miller Rocks gull colony, a colony where the vast majority (> 99%) of nesting gulls were California gulls (Table 2). Control tags sown on the colony prior to, during, and after the nesting season (n = 150) indicated that detection efficiency ranged from 74% to 90% for tags deposited between 1 April and 31 July (Table 3 and Appendix D, Figure D2). Deposition rates for Miller Rocks gulls were estimated to be 20% (95% c.i. = 15 - 25%; Table 4 and Appendix A).

Once adjusted for PIT tag detection efficiency and deposition rates, predation rates on smolts by gulls nesting at Miller Rocks ranged from a low of 0.8% (95% c.i. = 0.6 - 1.2%)

on Snake River spring/summer Chinook salmon to a high of 8.5% (95% c.i. = 5.6 - 12.3%) on Upper Columbia River steelhead (Table 9). Predation rates were also high on Snake River steelhead (4.7%; 95% c.i. = 3.4 – 6.6%) and Snake River sockeye (4.5%; 95% c.i. = 1.8 – 8.1%). Estimates of predation rates on upper Columbia River spring Chinook, Snake River fall Chinook, and upper Columbia River fall Chinook were < 2% per ESU, however (Table 9). These estimates of predation rates are similar to those observed at Miller Rocks in 2012, but are substantially higher than previously-reported estimates of gull predation rates that either did not incorporate corrections for PIT tag deposition rates (Evans et al. 2012) or used a deposition rate measured for Caspian terns (Lyons et al. 2011b). For example, using an assumed deposition rate of 70% (the best available data at the time), Lyons et al. (2011b) estimated gull predation rates of ≤ 2% on all salmonids ESUs, including steelhead, during 2007-2010. Deposition rate data collected in both 2012 and 2013 (see Appendix A), however, confirmed that the vast majority of salmonid PIT tags consumed by gulls are not deposited on-colony by gulls during the nesting season. As a result, predation rate estimates that are not adjusted or corrected for this low on-colony deposition rate underestimate impacts of gull predation on salmonids.

After adjustments for detection efficiency and deposition rates were made, estimates of smolt predation rates by gulls nesting on Miller Rocks were similar to or greater than comparable predation rate estimates by Caspian terns nesting at nearby islands in the Columbia Plateau region (Table 7). Due to the large size of the Miller Rocks gull colony (ca. 4,800 adults), estimates of per capita predation rates remain substantially less than those of Caspian terns nesting at either Crescent Island or Goose Island because these tern colonies are an order of magnitude smaller.

Miller Rocks is in close proximity to John Day Dam (< 10 km), a location where smolts may be particular vulnerable to avian predation (Zorich et al. 2011), and a location were dead or moribund smolts may be available to gulls. If gulls are disproportionately consuming dead or moribund PIT-tagged fish at dams or at other locations in the river, the impacts on salmonids would be lower than those implied by predation rate estimates due to compensatory mortality. None-the-less, predation rates on smolts by gulls nesting on Miller Rocks are comparable to those of tern and cormorant colonies in the Columbia Plateau region.

Blalock Islands Gulls – Following the nesting season, a total of 298 PIT-tagged smolts (Chinook, coho, sockeye, and steelhead, combined from all releases) from the 2013 migration year were recovered on a gull colony on an island in the Blalock Islands group (referred to as Anvil Island; Table 2), a colony where the majority (> 70%) of nesting gulls were California gulls (Figure 72). Control tags sown on the colony prior to, during, and after the nesting season (n = 150) indicated that detection efficiency ranged from 73% to 90% for tags deposited between 1 April and 31 July (Table 3 and Appendix D, Figure D2). Deposition rates for Blalock Islands gulls were estimated to be 15% (95% c.i. = 11 - 20%; Table 4 and Appendix A).

After adjustments for detection efficiency and deposition rate were made, predation rates by gulls nesting on Anvil Island were less than 1.2% for all salmonid ESUs/DPSs evaluated (Table 9), with predation rates on Snake River steelhead the highest (1.1%; 95% c.i. = 0.5 - 1.9%). Despite similarities in colony size and fish availability, predation rates on salmonids by gulls nesting on Anvil Island were significantly lower than those of gulls nesting on nearby Miller Rocks (Table 9), indicating substantial variation in predation rates on salmonids by gulls, depending on colony site. Similar to results from Miller Rocks gulls, steelhead populations and sockeye salmon were the most susceptible to Anvil Island gull predation. Overall, results indicate that gulls nesting on Anvil Island had a relatively small impact on smolt survival in 2013, especially compared to gulls nesting on Miller Rocks or Crescent Island (see below).

This was the first year when salmonid PIT tags from gull colonies in the Blalock Islands were recovered, and Anvil Island was not the only gull colony in the Blalock Islands group; a smaller colony of ring-billed gulls (ca. 1,110 adults) and California gulls (ca. 100 adults) was located on an island about 200 m from Anvil Island known as Straight Six Island (Figure 72). Cumulative impacts of waterbird species (i.e., gulls and Caspian terns) nesting at the Blalock Islands group may be larger than those implied by the estimate of predation rate by gulls nesting on Anvil Island, but may still be marginal (e.g., predation rates by Caspian terns nesting on Blalock Island were ≤ 0.1% per ESU/DPS; Table 9). It is worth noting, however, that gulls nesting on both Miller Rocks and Crescent Island (see below) – gull colonies where appreciable numbers of salmonid smolts were consumed in 2013 – are within foraging distance of the John Day Pool. As such, the cumulative impacts from gulls and other piscivorous waterbird species on smolt survival within the John Day Pool cannot be determined here because the location of smolt depredation is unknown.

Crescent Island Gulls – Following the nesting season, a total of 2,050 PIT-tagged smolts (Chinook, coho, sockeye, and steelhead, combined from all releases) from the 2013 migration year were recovered on the Crescent Island gull colony (Table 2), a colony where the vast majority (> 98%) of nesting gulls are California gulls (Figure 72). Control tags sown on the colony prior to, during, and after the nesting season (n = 150) indicated that detection efficiency ranged from 54% to 89% for tags deposited between 1 April and 31 July (Table 3 and Appendix D, Figure D2). Deposition rates for Crescent Island gulls were estimated to be 14% (95% c.i. = 10 - 20%; Table 4 and Appendix A).

Due to the presence of a Caspian tern colony on Crescent Island, some of the smolt PIT tags recovered on the Crescent Island gull colony were likely from smolts initially captured by Caspian terns and subsequently kleptoparasitized by gulls. Consequently, the total number of smolt PIT tags on the Crescent Island gull colony, and the resultant estimates of smolt predation rates, include smolts initially captured by Caspian terns and kleptoparasitized by gulls. Based on observed kleptoparasitism rates by gulls on Crescent Island in previous years (7 - 30%, depending on the salmonid species and year;

Adkins et al. 2011) and the fact that the majority (> 50%) of fish captured by Caspian terns during foraging bouts are immediately consumed by the tern (authors unpublished data) and not delivered to the colony where they could potentially be stolen by a gulls, the impact of kleptoparasitism on estimates of predation rates by Crescent Island gulls is likely less than 20%, but a precise value is unknown. Conceptually, the caveat regarding gull kleptoparasitism at Crescent Island is the same as the caveat regarding the consumption of dead or moribund fish at John Day Dam by gulls nesting at Miller Rocks, whereby the total amount of compensatory mortality in estimates of gull predation rates is unknown, but some proportion of the PIT-tagged fish consumed by gulls on Crescent Island were dead upon capture (killed by terns), resulting in compensatory mortality.

Once adjusted for PIT tag deposition rate and PIT tag detection efficiency, salmonid predation rates by gulls nesting at Crescent Island ranged from < 0.1% on Upper Columbia River spring Chinook to 6.1% (95% c.i. = 3.9 - 9.6%) on Upper Columbia River steelhead (Table 7). Estimates were also high for Snake River steelhead (4.8%; 95% c.i. = 3.2 - 4.7%) and to a lesser extent Snake River sockeye (1.2%; 95% c.i. = 0.1 - 3.1%), although sockeye salmon samples sizes were low, resulting in wide confidence intervals. Predation on all other ESUs, however, were ≤ 0.7% (Table 7).

In general, estimates of smolt predation rates by Crescent Island gulls in 2013 were similar to those in 2012 (BRNW 2013a), but are substantially higher than previously reported values, values that were either not corrected for deposition rates (Evans et al. 2012) or used a Caspian tern deposition estimates as a surrogate (Lyons et al. 2011b). Similar to predation rate estimates for gulls nesting on Miller Rocks, predation rates were equal to or greater than estimated predation rates by Caspian terns nesting on Crescent Island (Table 7). The per capita impact of Crescent Island gulls on smolt survival, however, remained considerably less than those of Crescent Island and Goose Island Caspian terns because the size of the Crescent Island gull colony was much larger (ca. 5,710 adults; Figure 72) in 2013.

Island 20 Gulls – Following the 2013 nesting season, a total of 478 PIT-tagged smolts (Chinook, coho, sockeye, and steelhead, combined from all releases) from the 2013 migration year were recovered on the gull colony on Island 20 (Table 2), a large colony that consisted of both California gulls (ca. 8,980) and ring-billed gulls (ca. 5,060; Figure 72). Control PIT tags sown on the colony prior to, during, and after the nesting season (n = 150) indicated that detection efficiency ranged from 65% to 91% for tags deposited between 1 April and 31 July (Table 3 and Appendix D, Figure D2). Deposition rates for Island 20 gulls were estimated to be 15% (95% c.i. = 10 - 20%; Table 4 and Appendix A).

After adjustments for detection efficiency and deposition rate were made, smolt predation rates by gulls nesting on Island 20 were < 1.4% on all ESUs/DPSs evaluated (Table 7). Predation rates on Upper Columbia River steelhead were the highest at 1.3% (95% c.i. = 0.5 - 2.4%). Similar to results from other gull colonies, steelhead populations were more susceptible to gull predation relative to their availability than were salmon populations. Relative to the three other gull colonies evaluate in 2013, predation rates on salmonids by gulls nesting on Island 20 were, on average, the lowest observed (Table 7 and Table 9). This result is somewhat surprising given the large size of the Island 20 gull colony, with 14,000 adult gulls counted on-colony during the peak the nesting season. Results from all four gull colonies evaluated in 2013 indicate that large differences in the diet composition and foraging behavior of gulls exist between colonies in the Columbia Plateau region.

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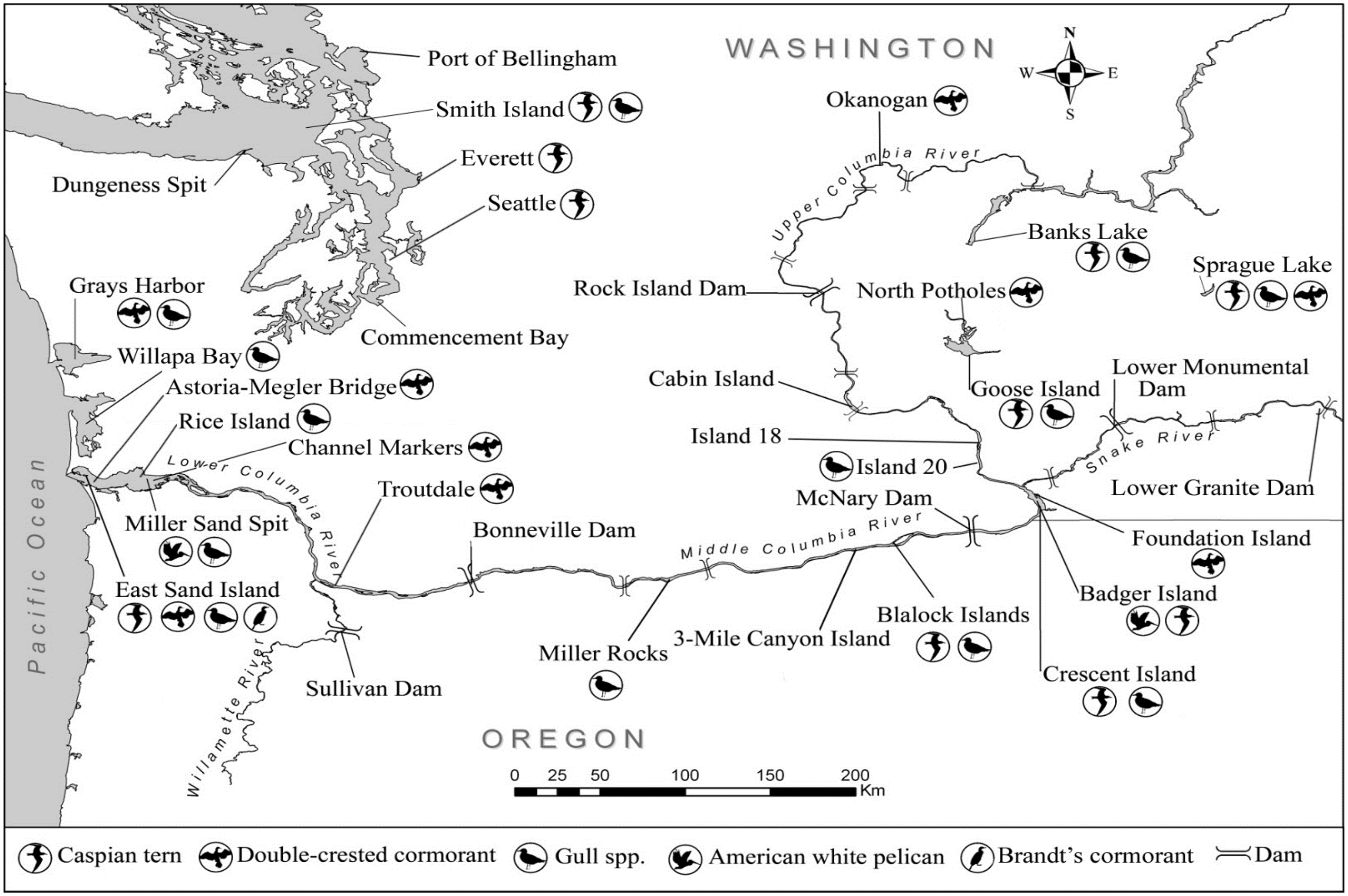
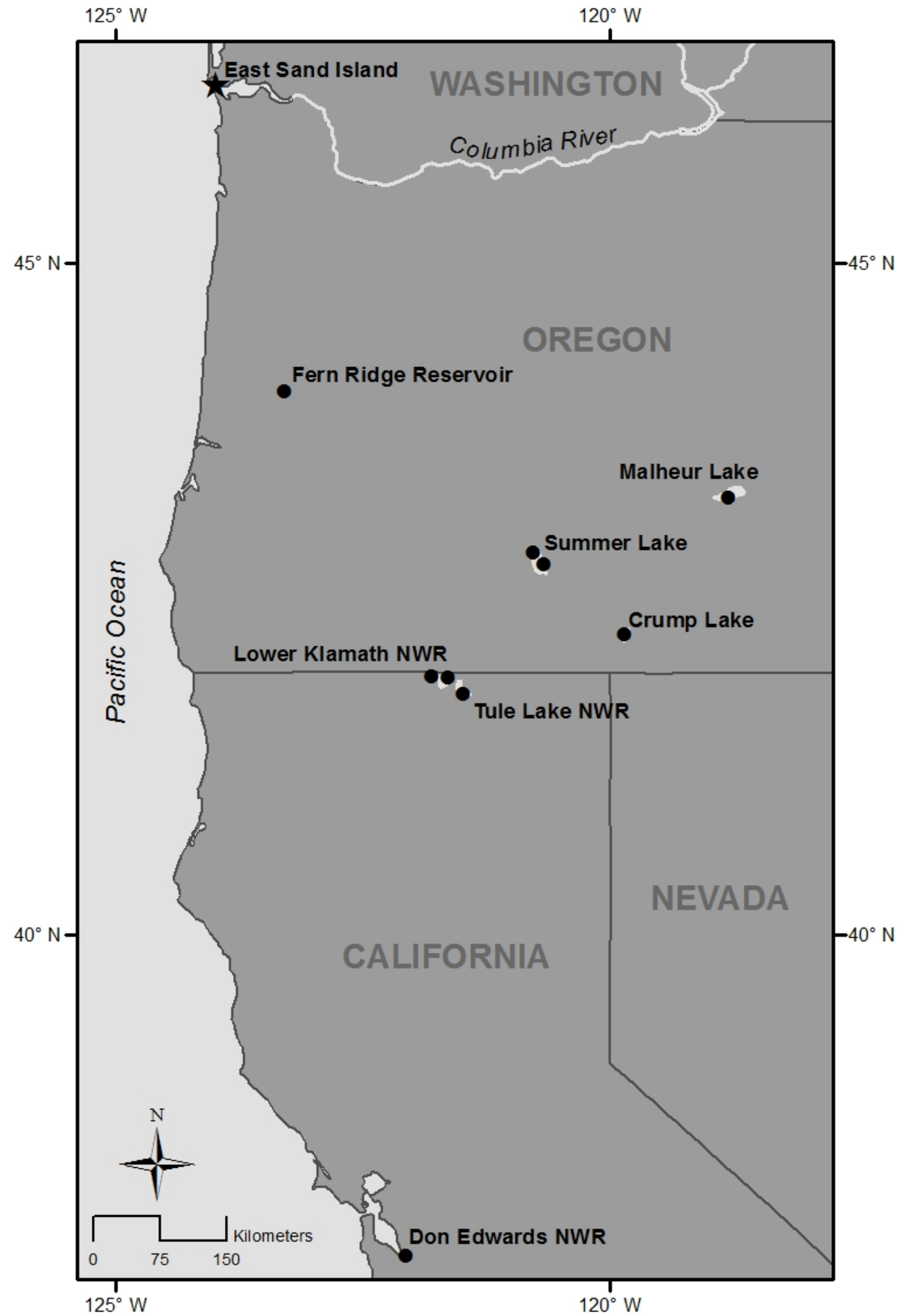
	Funding Contribution by Agency			
	BPA	USACE Portland District	USACE Walla Walla District	Grant Co. PUD
Caspian Terns				
1.1. Preparation and Modification of Nesting Habitat				
1.1.1. Columbia River Estuary		**x**		
1.1.2. Interior OR and Northeastern CA		**x**		
1.2. Colony Size and Productivity				
1.2.1. Columbia River Estuary	**x**			
1.2.2. Columbia Plateau			**x**	**x**
1.2.3. Coastal Washington		**x**		
1.2.4. Interior OR and Northeastern CA		**x**		
1.3. Diet Composition and Salmonid Consumption				
1.3.1. Columbia River Estuary	**x**			
1.3.2. Columbia Plateau			**x**	**x**
1.3.3. Coastal Washington				
1.3.4. Interior OR and Northeastern CA		**x**		
1.4. Predation Rates Based on PIT Tag Recoveries				
1.4.1. Columbia River Estuary	**x**	**x**		
1.4.2. Columbia Plateau			**x**	**x**
1.4.3. Coastal Washington				

	Funding Contribution by Agency			
	BPA	USACE Portland District	USACE Walla Walla District	Grant Co. PUD
Caspian Terns (cont.)				
1.4.4. Interior OR and Northeastern CA		**x**		
1.5. Color Banding and Band Re-sightings				
1.5.1. Columbia River Estuary	**x**			
1.5.2. Columbia Plateau			**x**	**x**
1.5.3. Coastal Washington				
1.5.4. Interior OR and Northeastern CA		**x**		
Double-crested Cormorants				
2.1. Nesting Distribution and Colony Size				
2.1.1. Columbia River Estuary	**x**	**x**		
2.1.2. Columbia Plateau			**x**	
2.1.3. Coastal Washington		**x**		
2.1.4. Interior OR and Northeastern CA		**x**		
2.2. Nesting Success				
2.2.1. Columbia River Estuary	**x**	**x**		
2.2.2. Columbia Plateau			**x**	
2.2.3. Coastal Washington		**x**		
2.2.4. Interior OR and Northeastern CA		**x**		
2.3. Diet Composition and Salmonid Consumption				
2.3.1. Columbia River Estuary	**x**	**x**		
2.3.2. Columbia Plateau				
2.3.3. Coastal Washington				
2.3.4. Interior OR and Northeastern CA				
2.4. Predation Rates Based on PIT Tag Recoveries				
2.4.1. Columbia River Estuary	**x**	**x**		
2.4.2. Columbia Plateau				
2.5. Color Banding		**x**		
2.6. Management Feasibility Study		**x**		

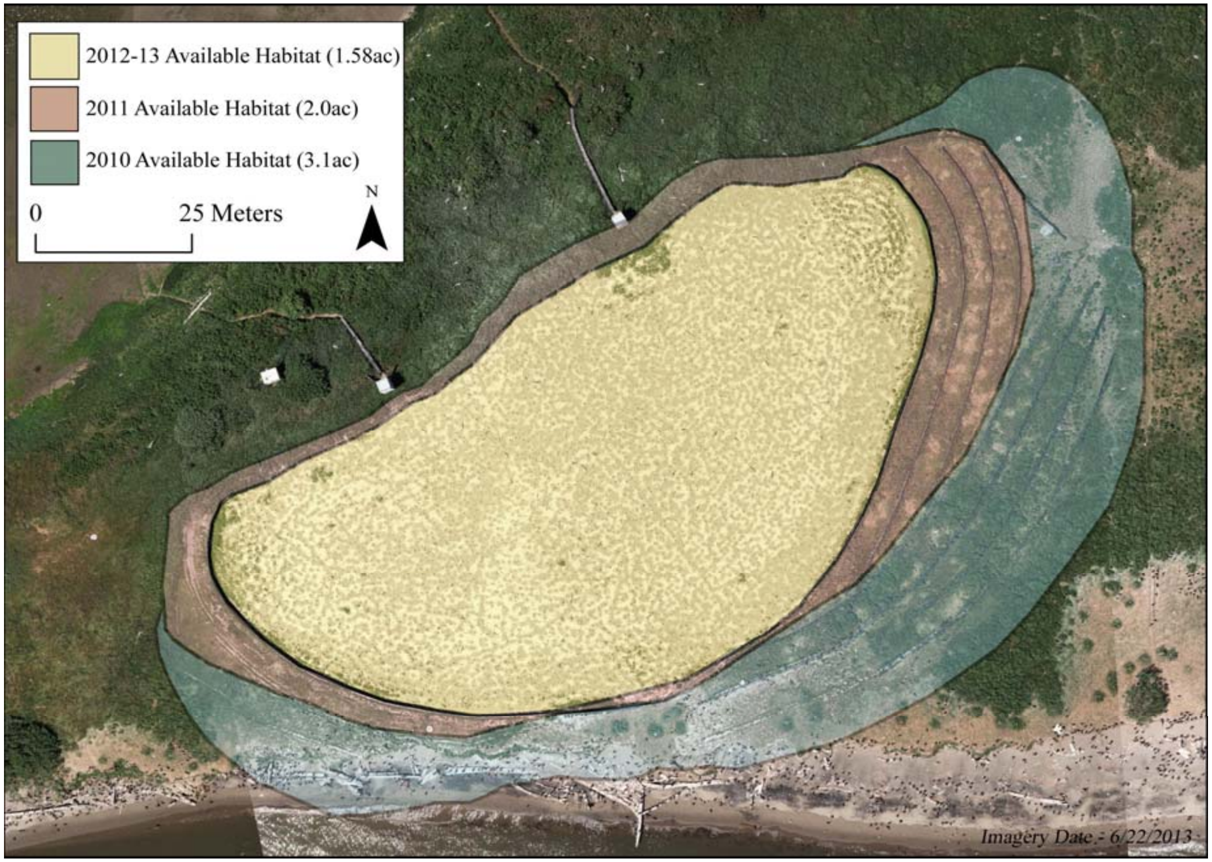
	Funding Contribution by Agency			
	BPA	USACE Portland District	USACE Walla Walla District	Grant Co. PUD
Other Piscivorous Waterbirds				
3.1. Distribution				
3.1.1. Columbia River Estuary	**x**			
3.1.2. Columbia Plateau			**x**	
3.1.3. Coastal Washington				
3.1.4. Interior Oregon and Northeastern California		**x**		
3.2. Diet Composition				
3.2.1. Columbia River Estuary				
3.2.2. Columbia Plateau				
3.2.3. Coastal Washington				
3.2.4. Interior Oregon and Northeastern California				
3.3. Predation Rates Based on PIT Tag Recoveries				
3.3.1. Columbia River Estuary	**x**			
3.3.2. Columbia Plateau			**x**	
Foraging Behavior and Dispersal Patterns of Caspian Terns Nesting on Goose Island				**x**



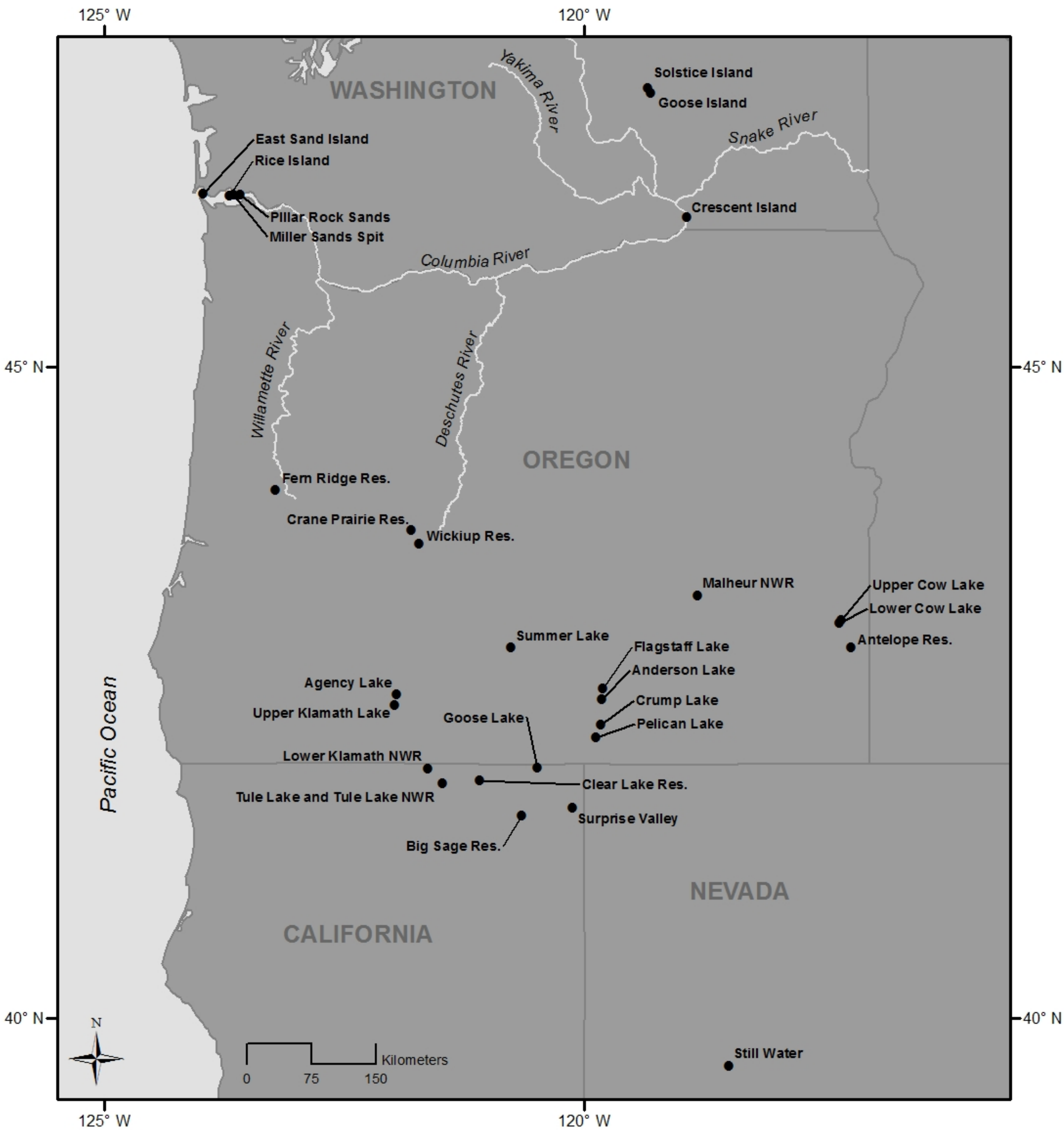
Map 1. Study area in the Columbia River basin and coastal Washington showing the locations of active and former breeding colonies of piscivorous colonial waterbirds mentioned in this report. A Caspian tern colony was also discovered on a warehouse rooftop at the Fraser River Terminal near Richmond, British Columbia (not shown on map).



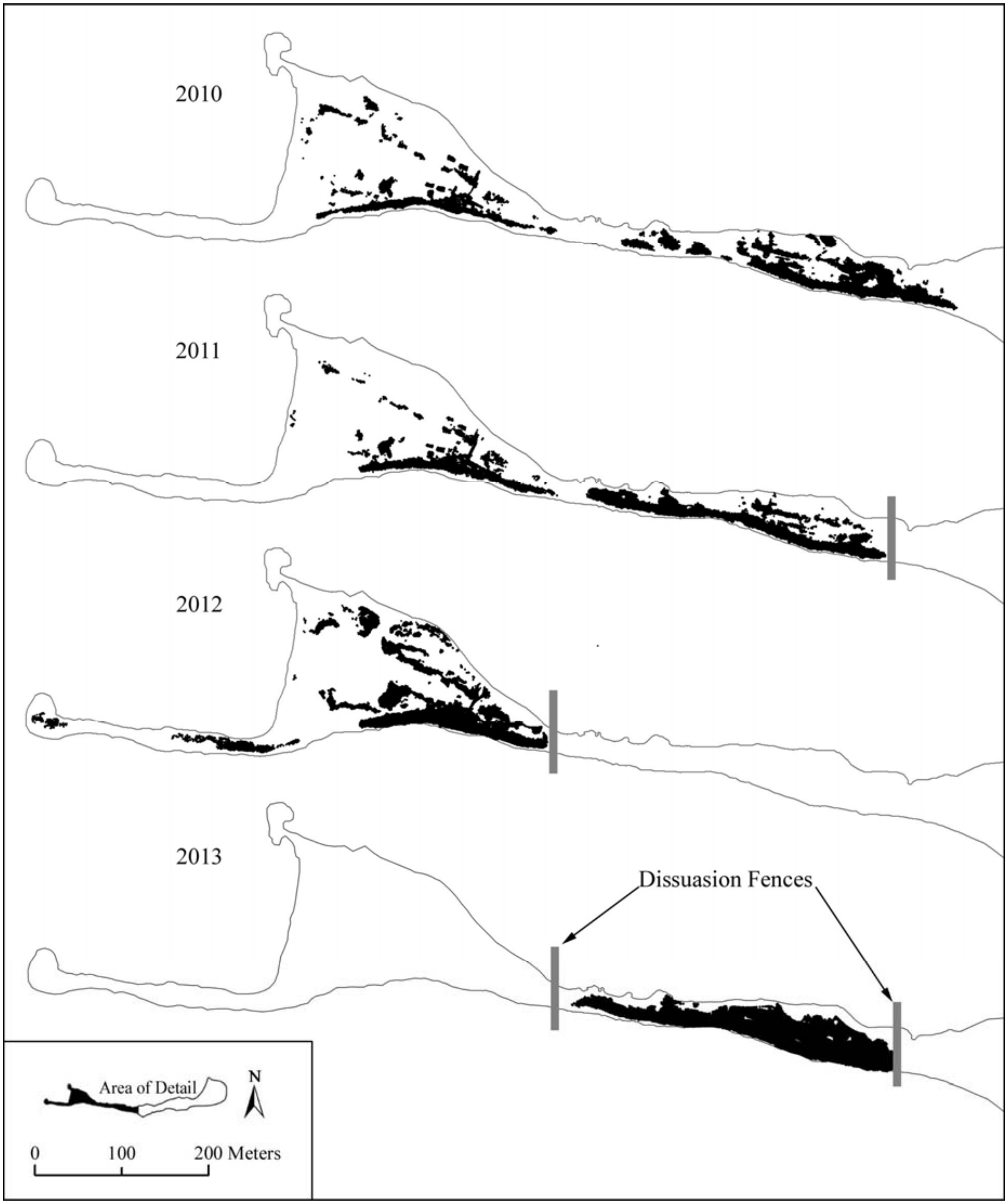
Map 2. Locations of recently-built and proposed Corps-constructed islands for Caspian tern nesting as part of the federal agencies' Caspian Tern Management Plan for the Columbia River estuary (USFWS 2005, 2006).



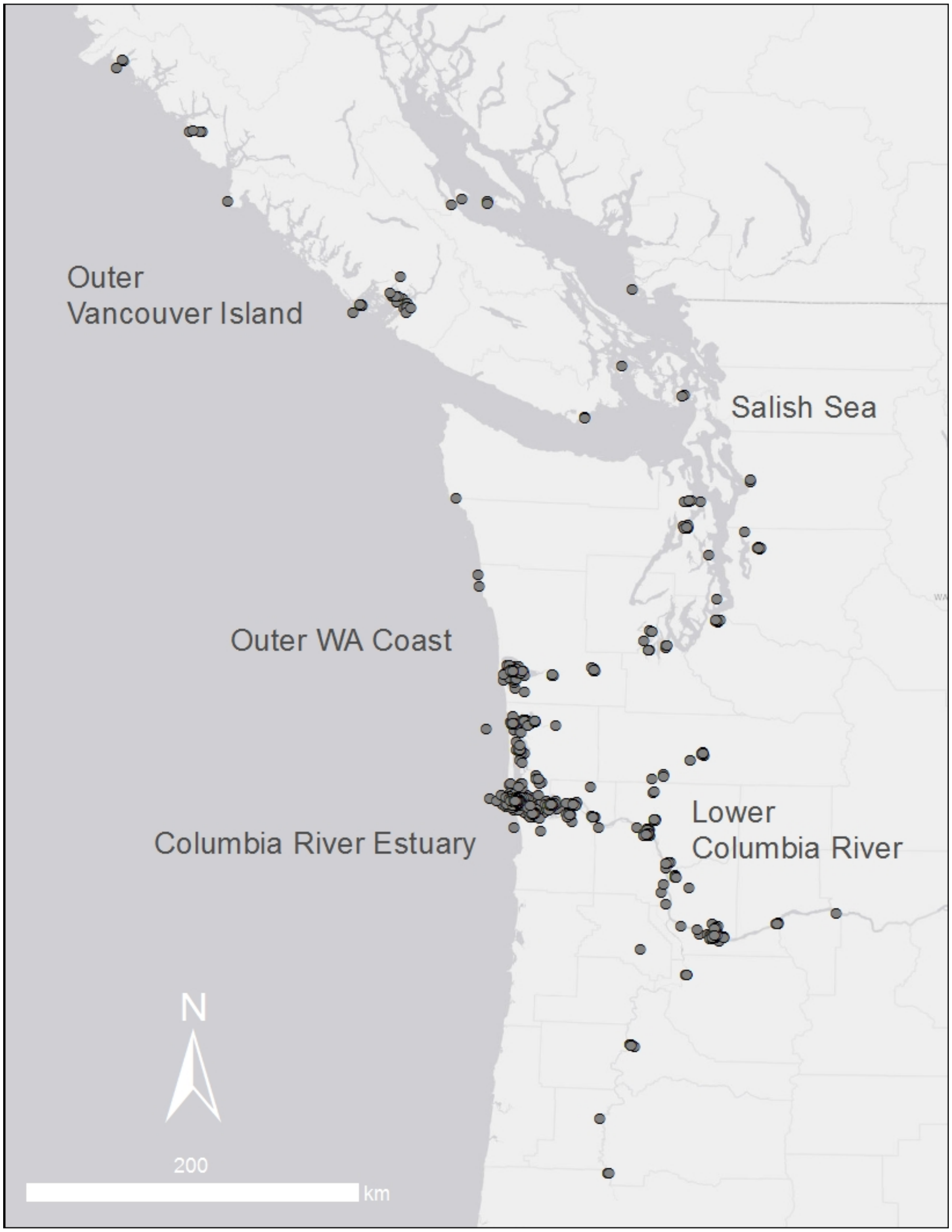
Map 3. Nesting habitat prepared for Caspian terns on the eastern end of East Sand Island in the Columbia River estuary during 2010-2013. Silt fencing was erected on a portion of the nesting habitat used by terns in 2010-2011 to further reduce the amount of nesting habitat made available to Caspian terns during 2012-2013 (see text for details). During 2010-2012, Caspian terns nested only in the designated tern colony area at the eastern end of East Sand Island (shown here), and not elsewhere on the island. In 2013, terns attempted to nest outside the designated colony area (on the beaches surrounding the colony area to the north, east, and south) but were unsuccessful in raising young at those locations due to inundation of nests during high high-tide events.



Map 4. Study area in interior Oregon, northeastern California, and southern Washington, with locations of piscivorous waterbird colonies mentioned in this report.



Map 5. Distribution of cormorant nests on western East Sand Island in the Columbia River estuary during the 2010-2013 breeding seasons. During 2010-2013, cormorants nested only on the western half of East Sand Island (shown here) and not elsewhere on the island.



Map 6. Off-colony detection locations of satellite-tagged double-crested cormorants from East Sand Island in the Columbia River estuary during the breeding season (April-September) in 2013. Birds were captured in April 2013 for tagging on a portion of the colony where cormorants were prevented from nesting, east and west of the dissuasion fences (i.e., dissuasion area; see Map 5). Dispersal from East Sand Island was to 5 regions: Columbia River Estuary, Lower Columbia River, Outer Washington Coast, Salish Sea, and Outer Vancouver Island.

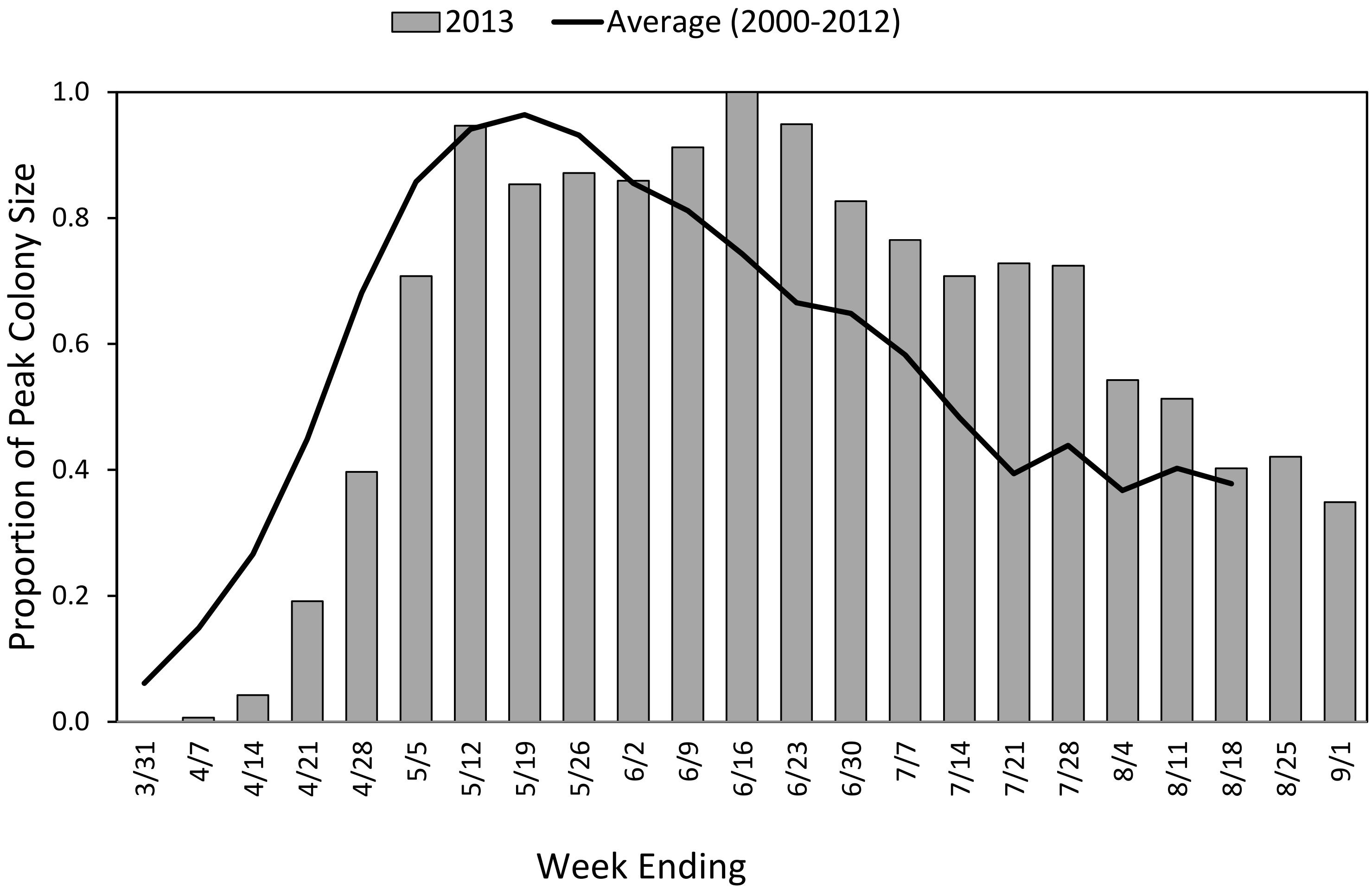


Figure 1. Weekly estimates from the ground of the number of adult Caspian terns on the East Sand Island colony during the 2013 breeding season, relative to peak colony attendance determined from counts of aerial photography taken late in the incubation period.

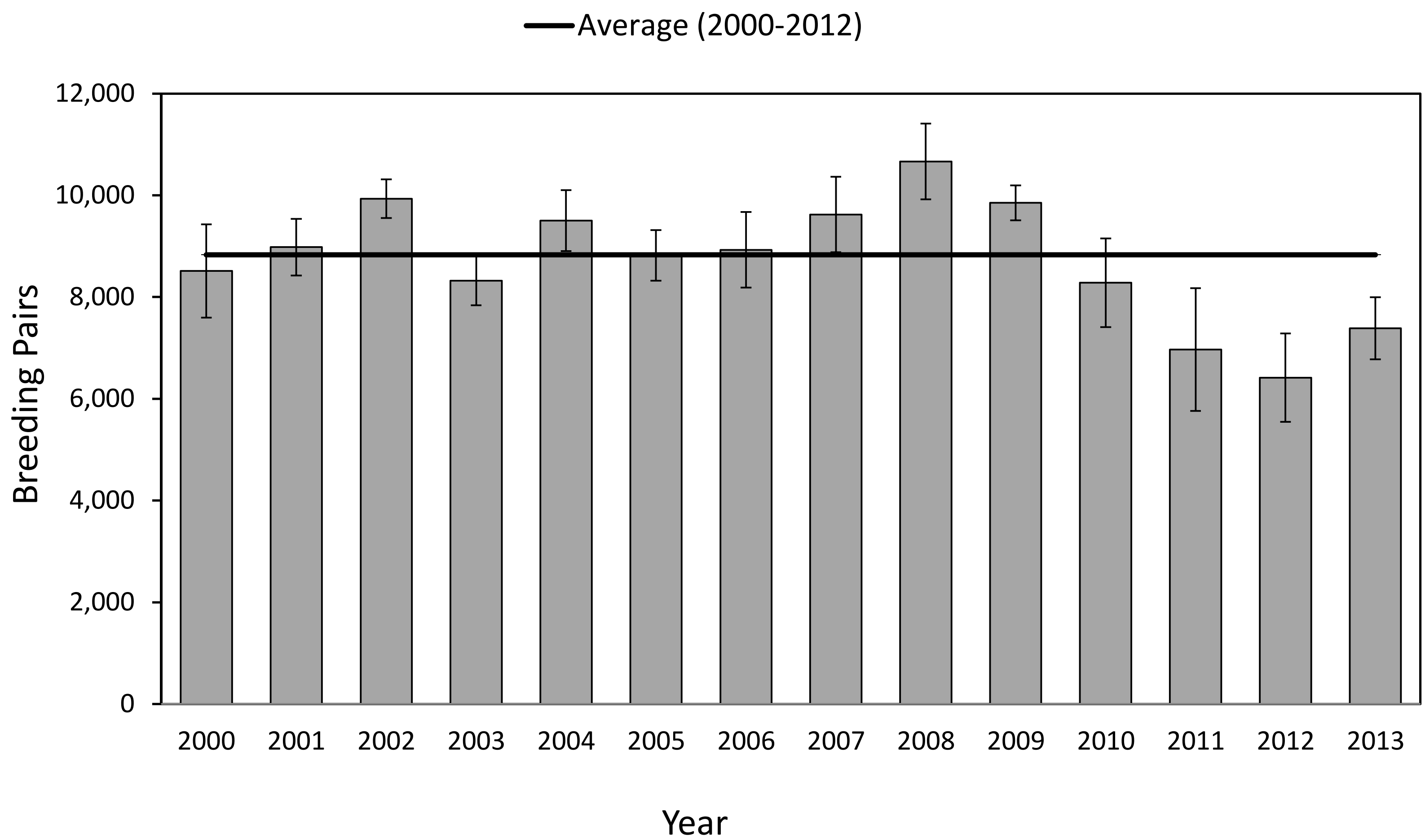


Figure 2. Caspian tern colony size (number of breeding pairs) on East Sand Island in the Columbia River estuary during 2000-2013. The error bars represent 95% confidence intervals for the number of breeding pairs.

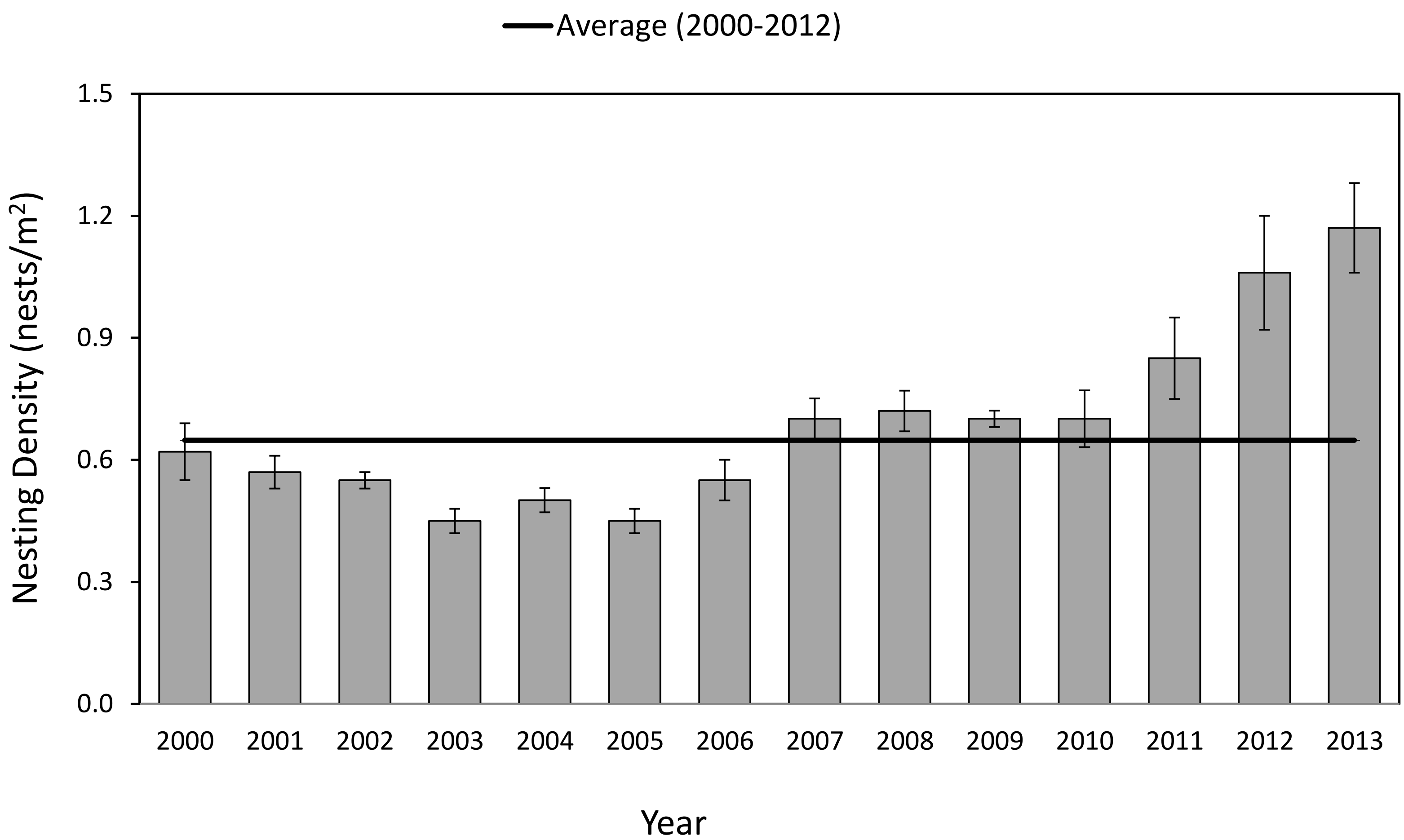


Figure 3. Caspian tern nesting density at the breeding colony on East Sand Island, Columbia River estuary during 2000-2013. The error bars represent 95% confidence intervals for nesting density (error estimate not available for 2011 and based on 2012 error estimate).

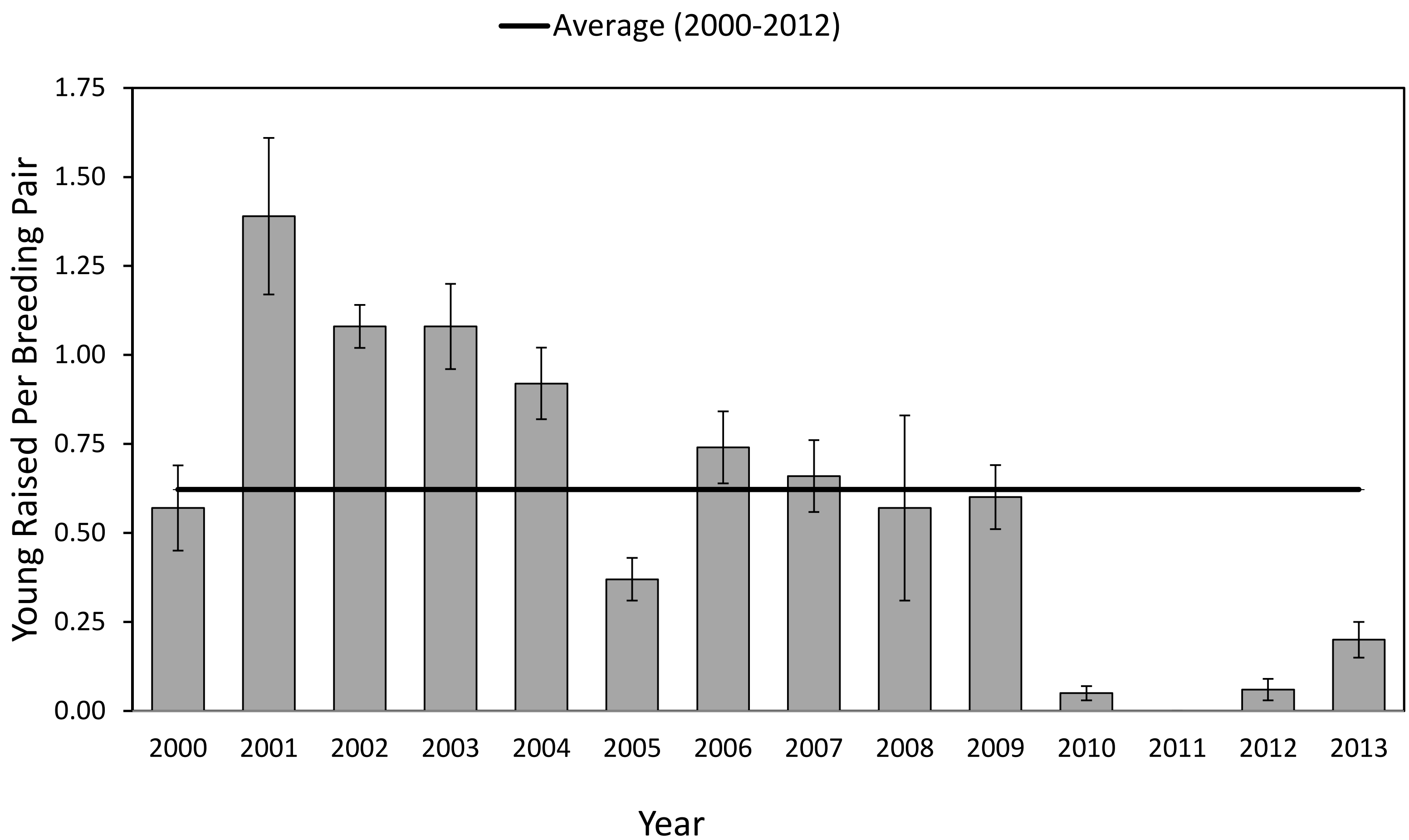


Figure 4. Caspian tern nesting success (average number of young raised per breeding pair) at the breeding colony on East Sand Island in the Columbia River estuary during 2000-2013. The error bars represent 95% confidence intervals.

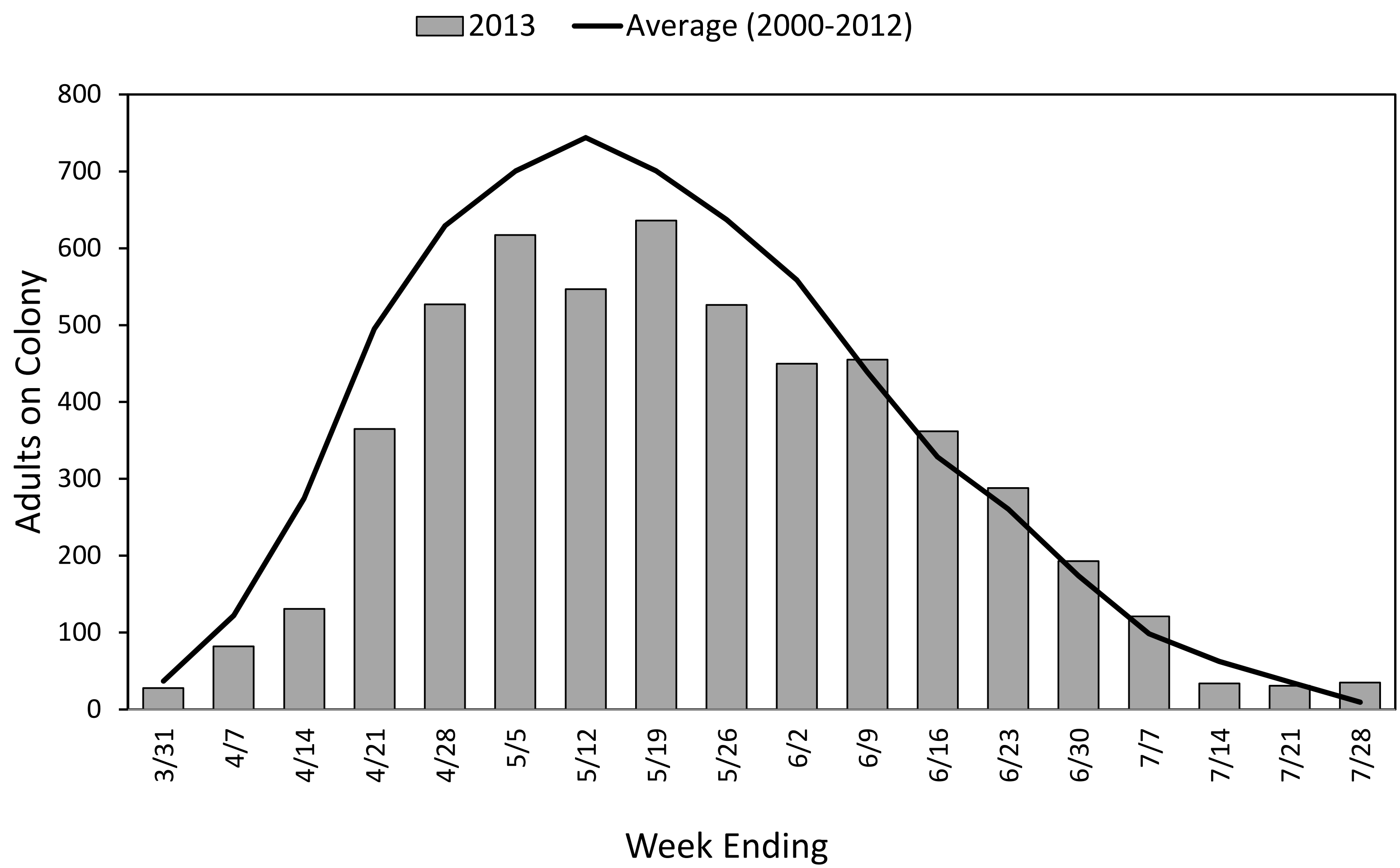


Figure 5. Estimates from the ground of the number of adult Caspian terns on the Crescent Island breeding colony in the mid-Columbia River, by week during the 2013 breeding season.

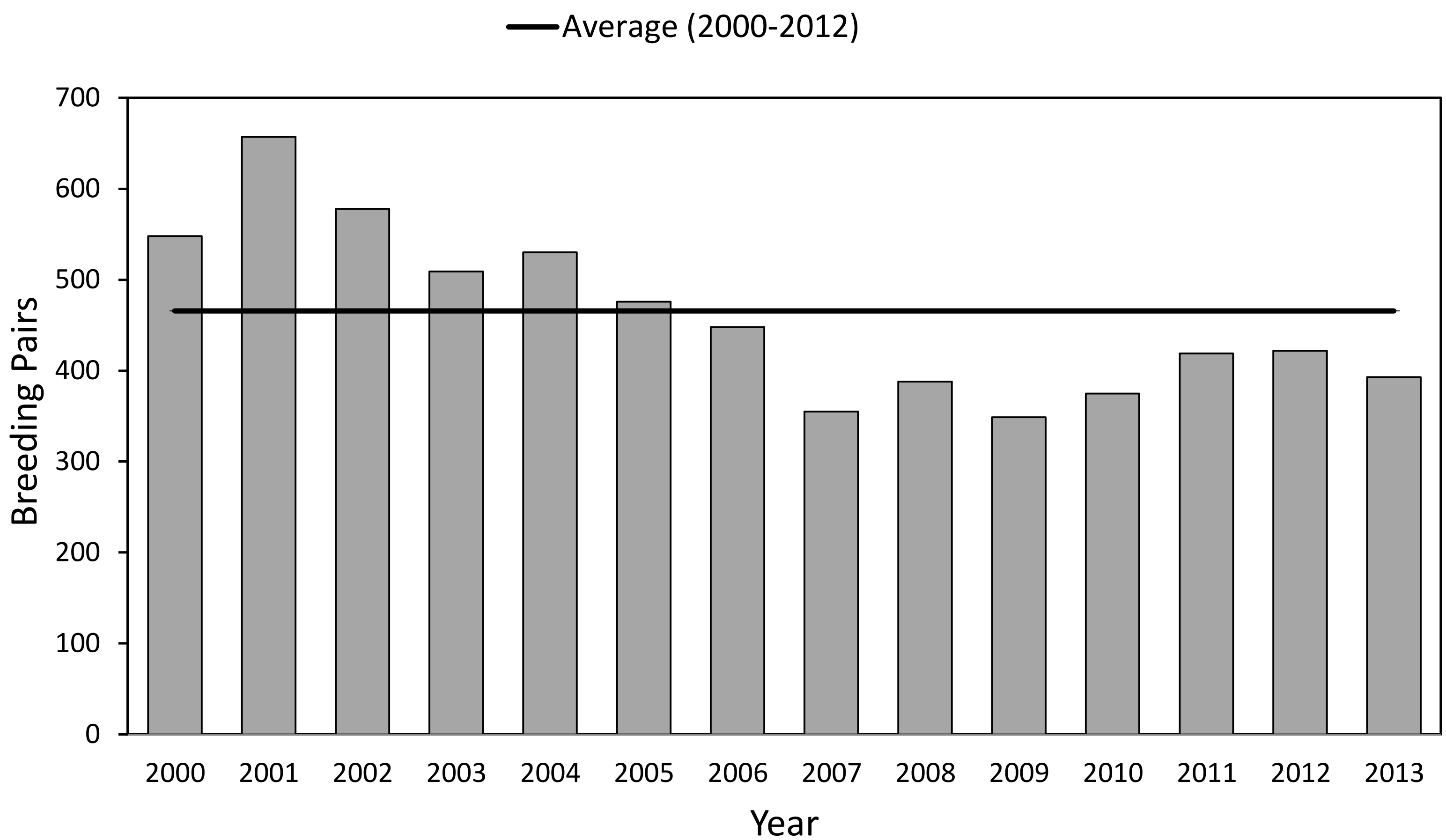


Figure 6. Size of the Caspian tern breeding colony (number of breeding pairs) on Crescent Island in the mid-Columbia River during 2000-2013.

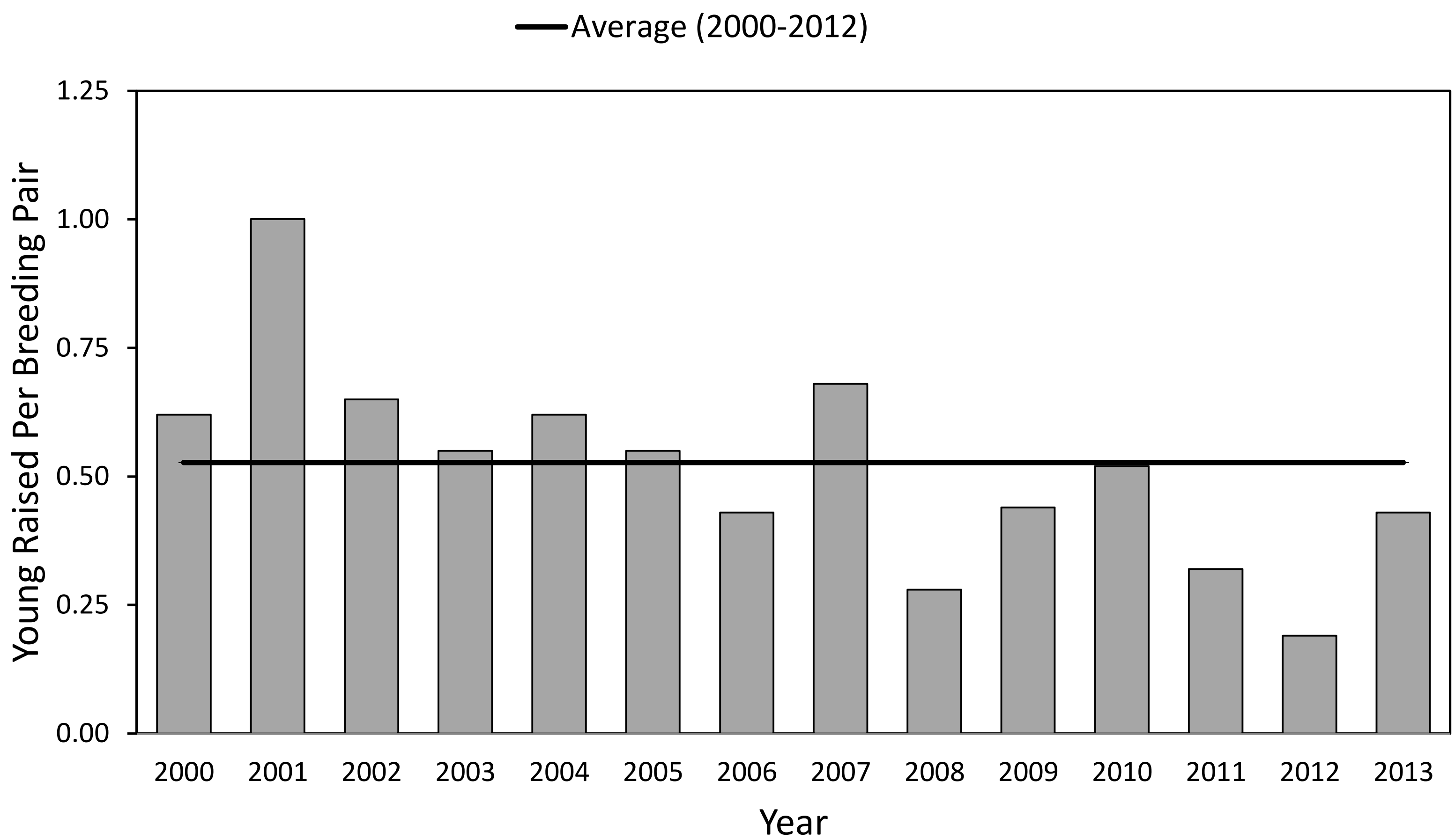


Figure 7. Nesting success of Caspian terns (average number of young raised per breeding pair) at the breeding colony on Crescent Island in the mid-Columbia River during 2000-2013.

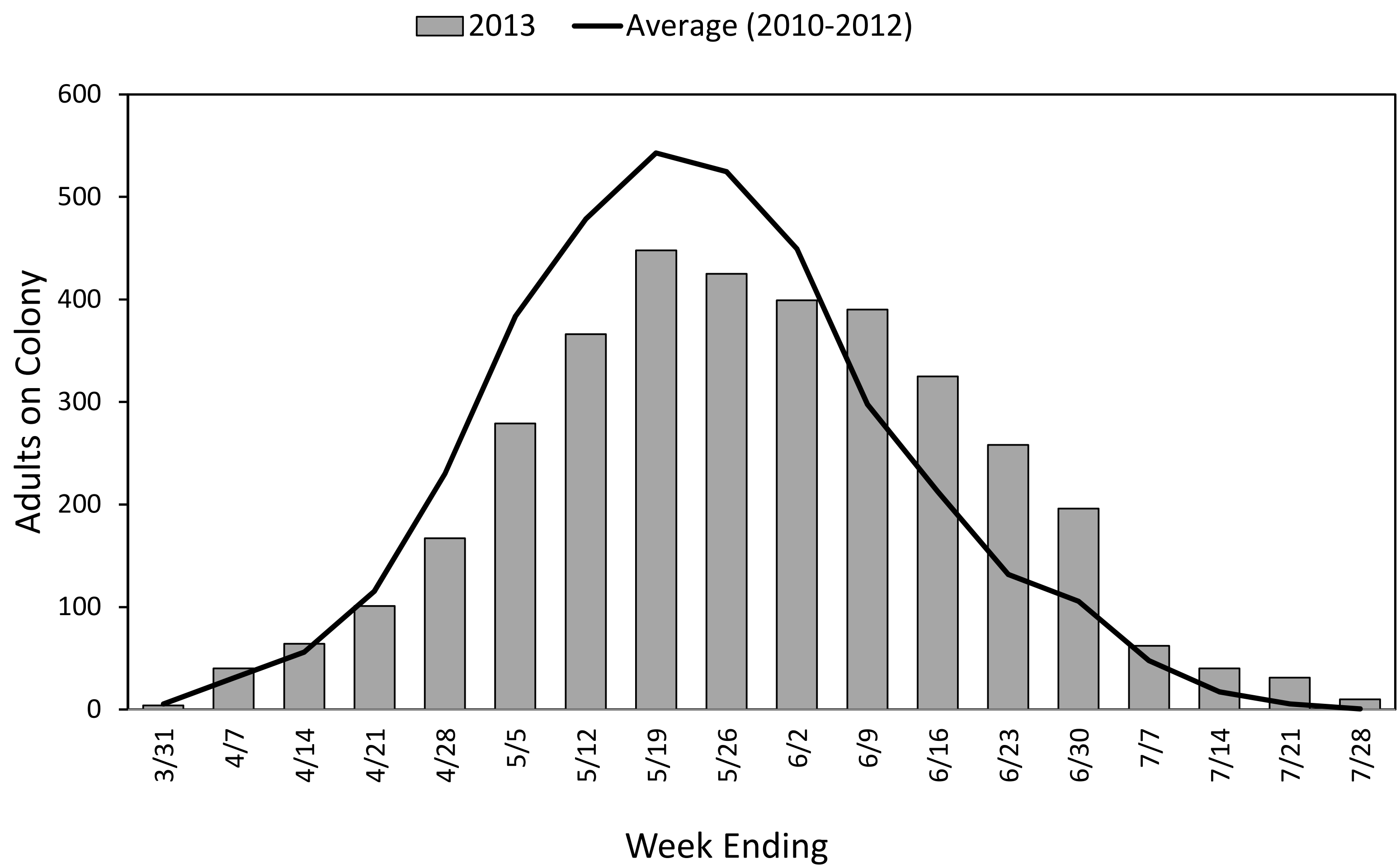


Figure 8. Estimates from the ground of the number of adult Caspian terns at the breeding colony on Goose Island in Potholes Reservoir, by week during the 2013 breeding season.

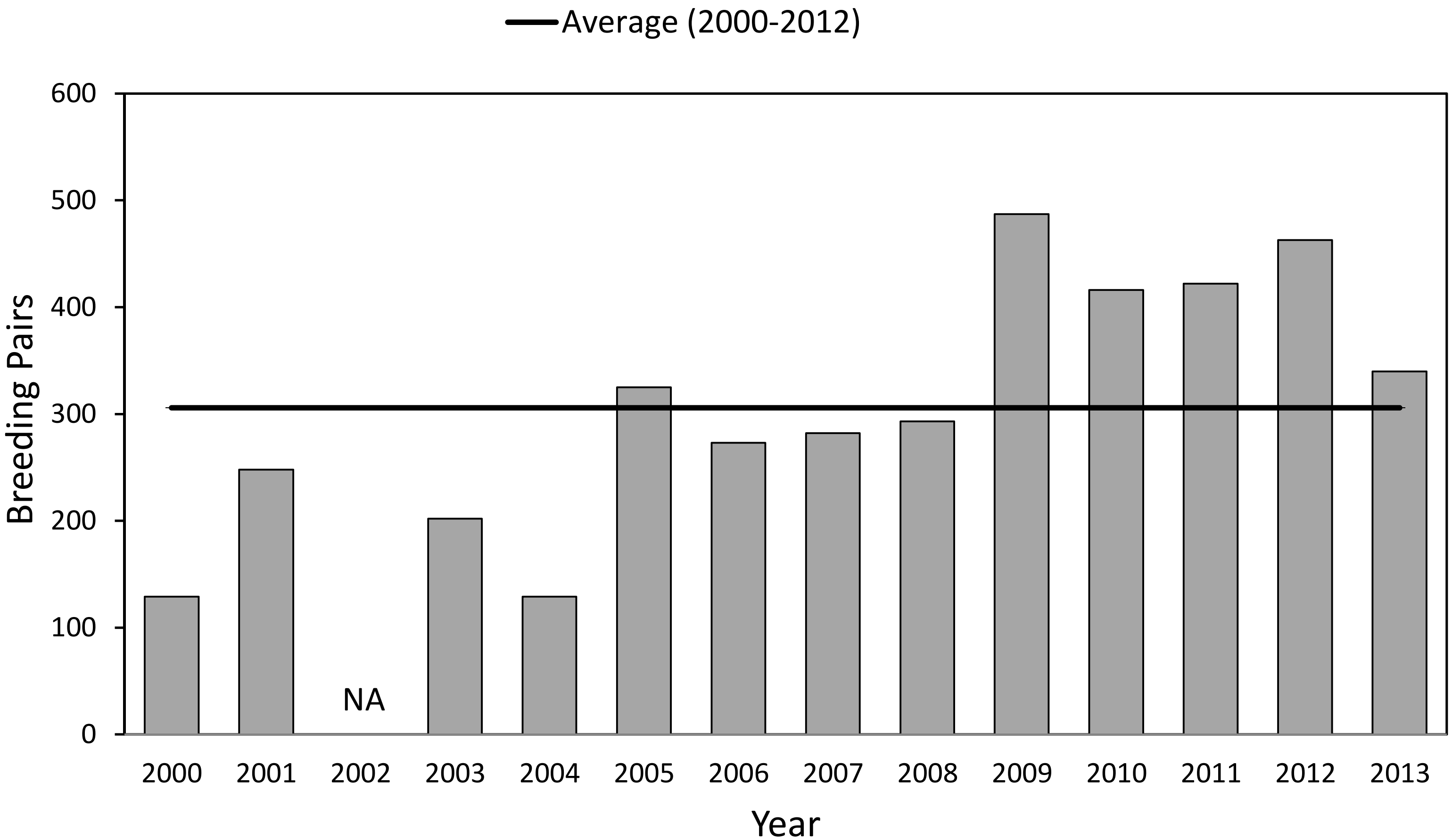


Figure 9. Size of the Caspian tern breeding colony (number of breeding pairs) at Potholes Reservoir during 2000-2013. The colony was located on Solstice Island during 2000-2004, and on Goose Island from 2004 to present. Colony size in 2002 is not known.

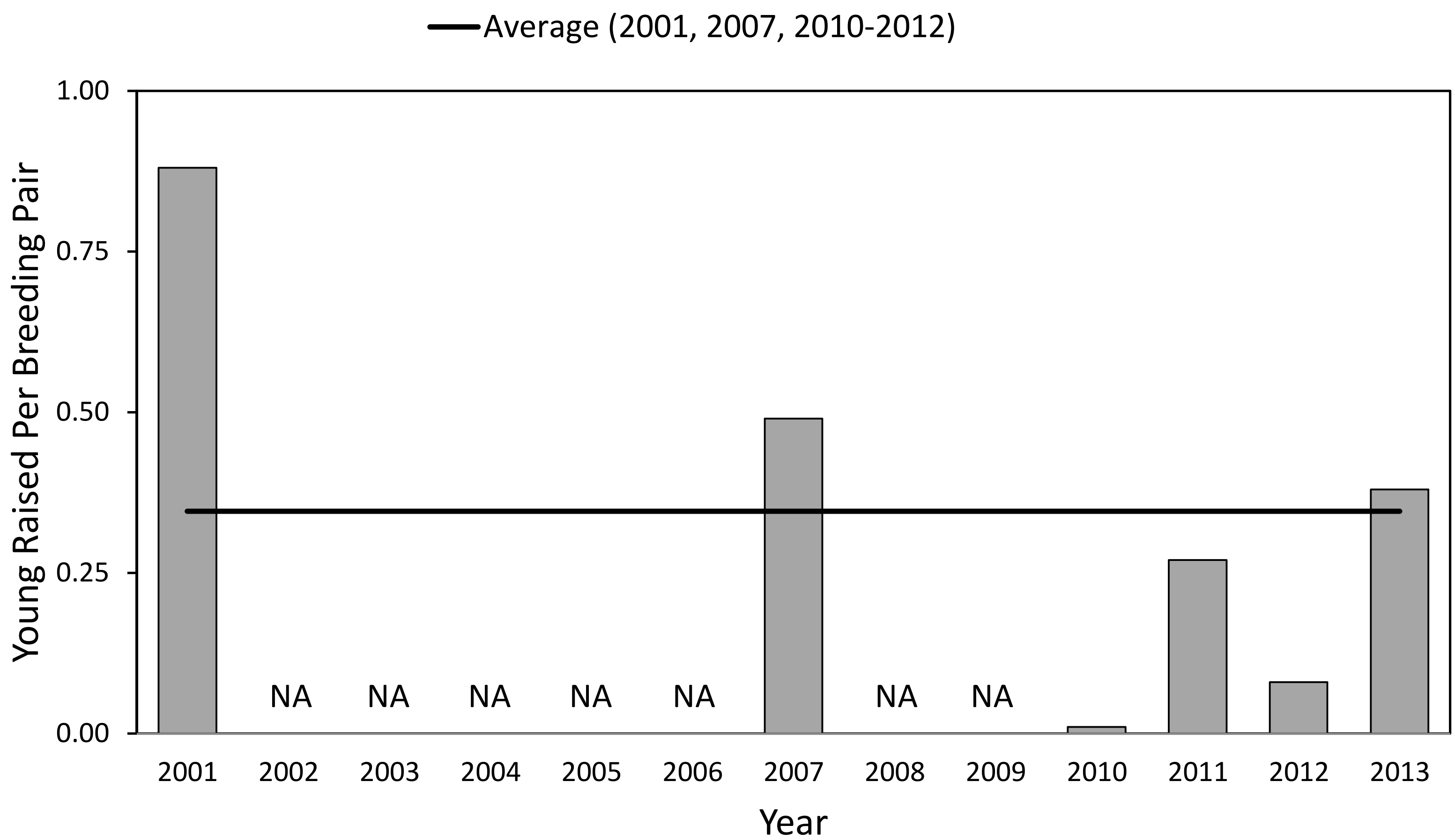


Figure 10. Caspian tern nesting success (average number of young raised per breeding pair) at the breeding colony at Potholes Reservoir during 2001-2013. The colony was on Solstice Island during 2001-2004, and on Goose Island from 2004 to present. Nesting success during 2002-2006 and 2008-2009 is not known.

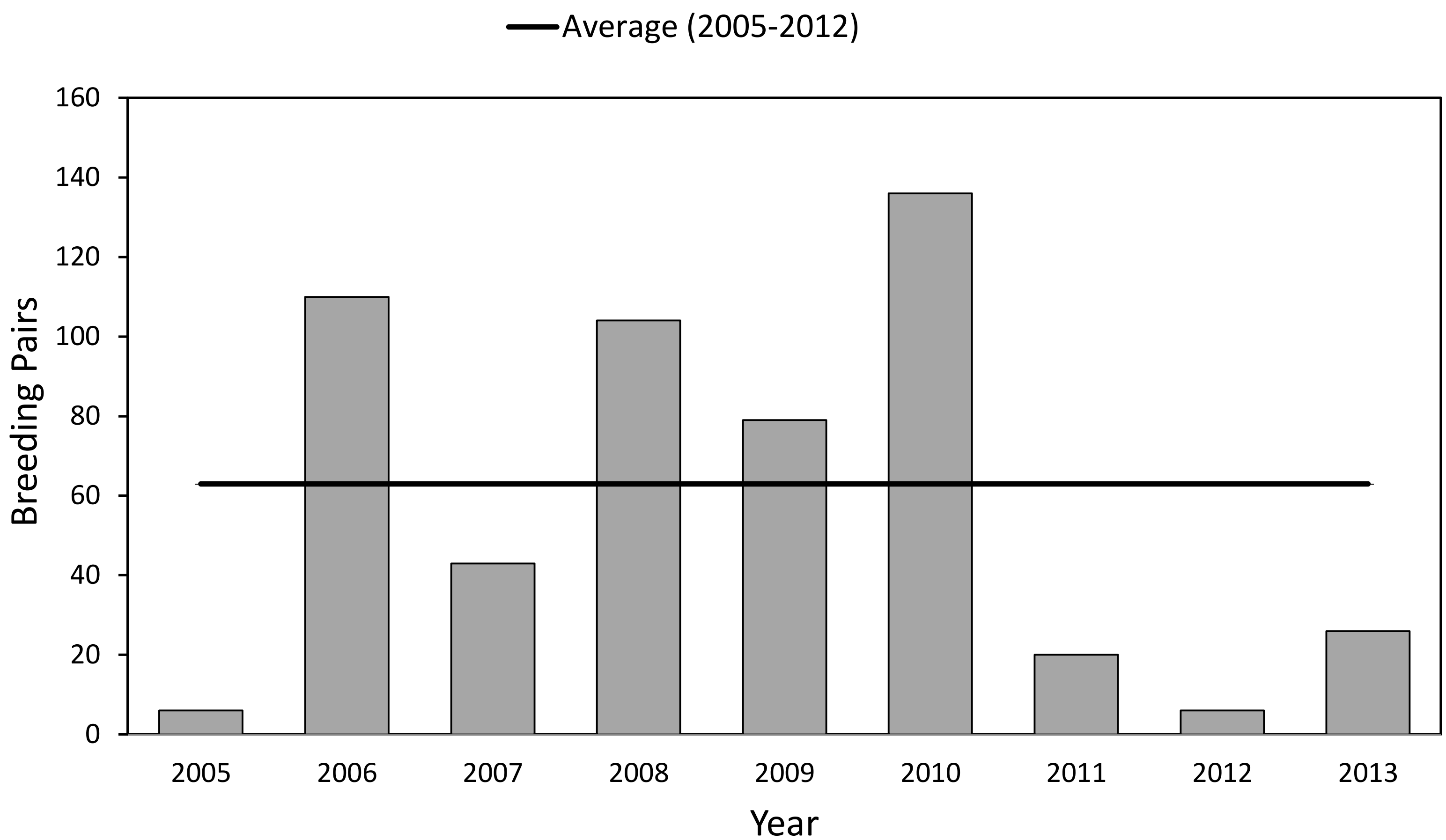


Figure 11. Size of the Caspian tern breeding colony (number of breeding pairs) at the Blalock Islands in the mid-Columbia River during 2005-2013.

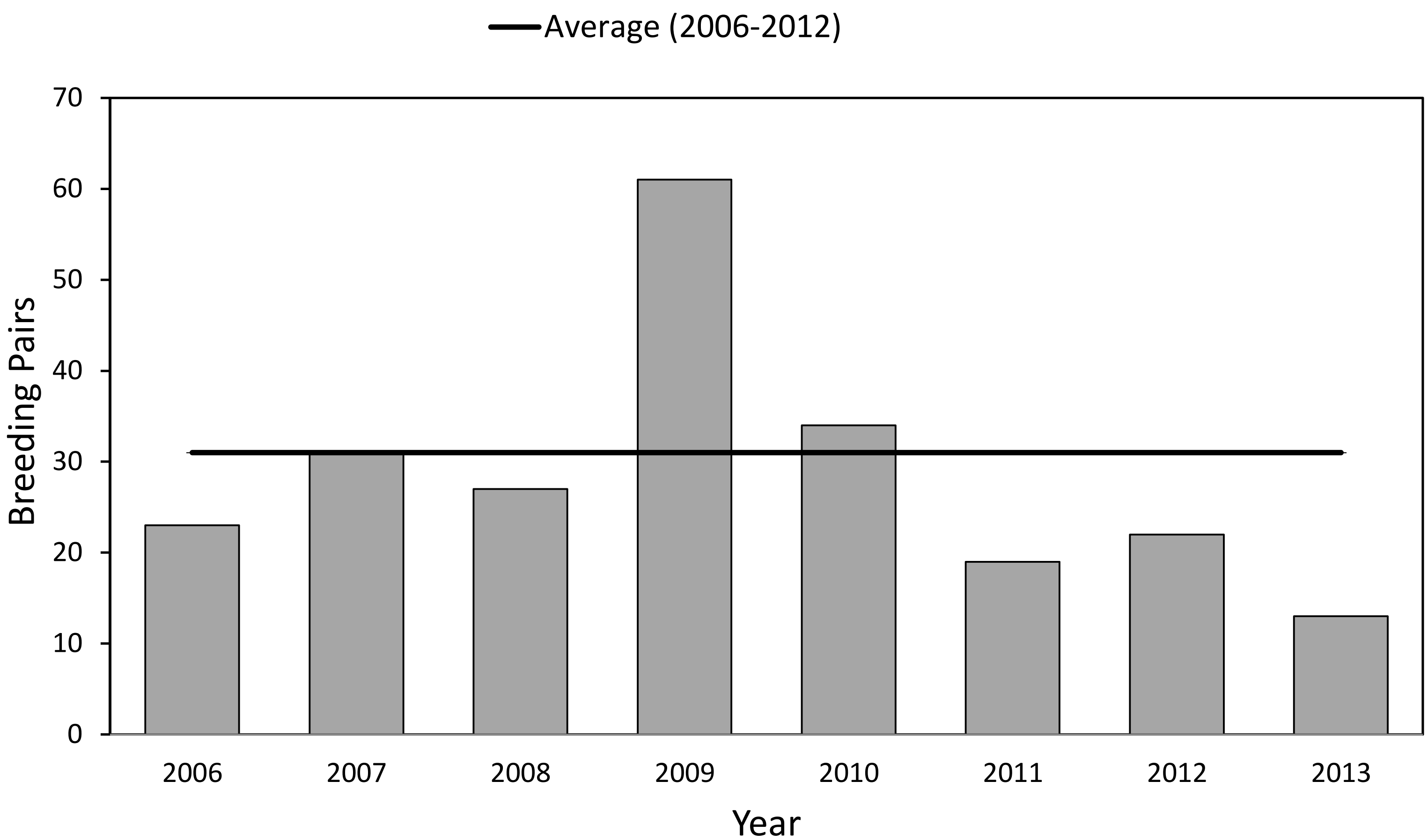


Figure 12. Size of the Caspian tern breeding colony (number of breeding pairs) at Twining Island in Banks Lake during 2006-2013. In 2005, Caspian terns nested on two islands in Banks Lake (Twining and Goose islands), and colony size was estimated to be less than 10 breeding pairs at each site.

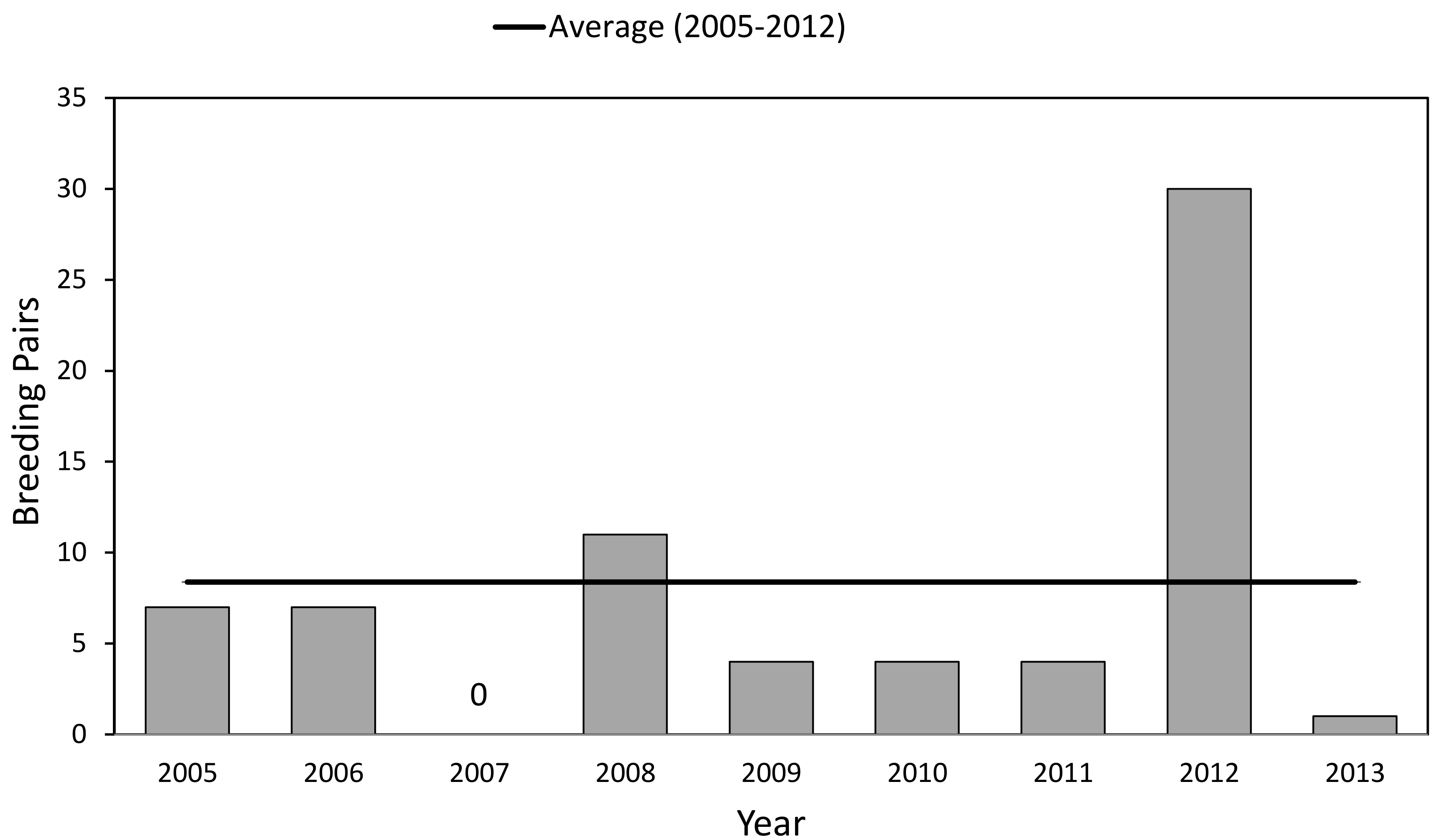


Figure 13. Size of the Caspian tern breeding colony (number of breeding pairs) at Harper Island in Sprague Lake during 2005-2013. Caspian terns did not attempt to nest on Harper Island in 2007.

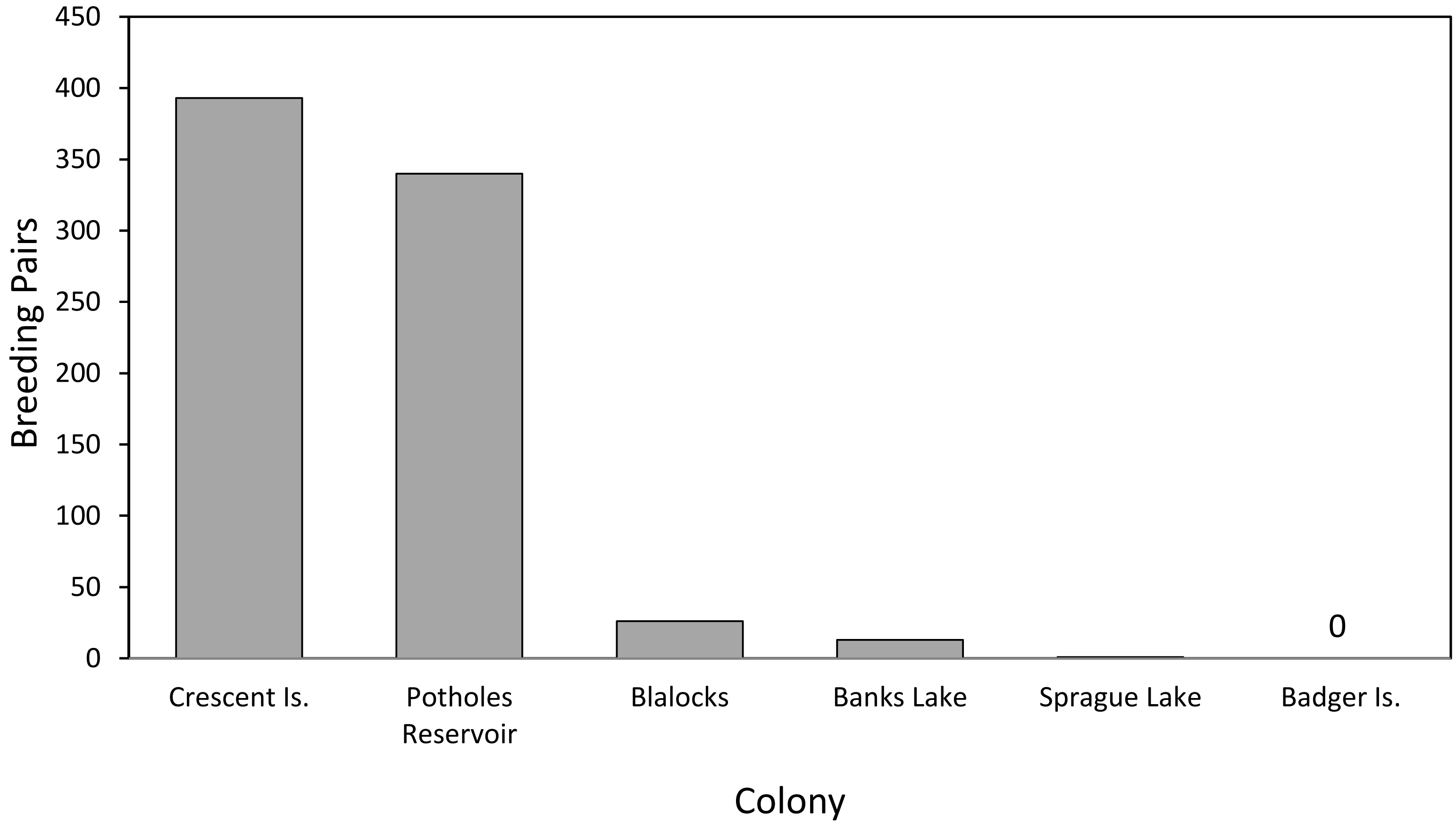


Figure 14. Sizes of Caspian tern breeding colonies (numbers of breeding pairs) in the Columbia Plateau region during the 2013 breeding season.

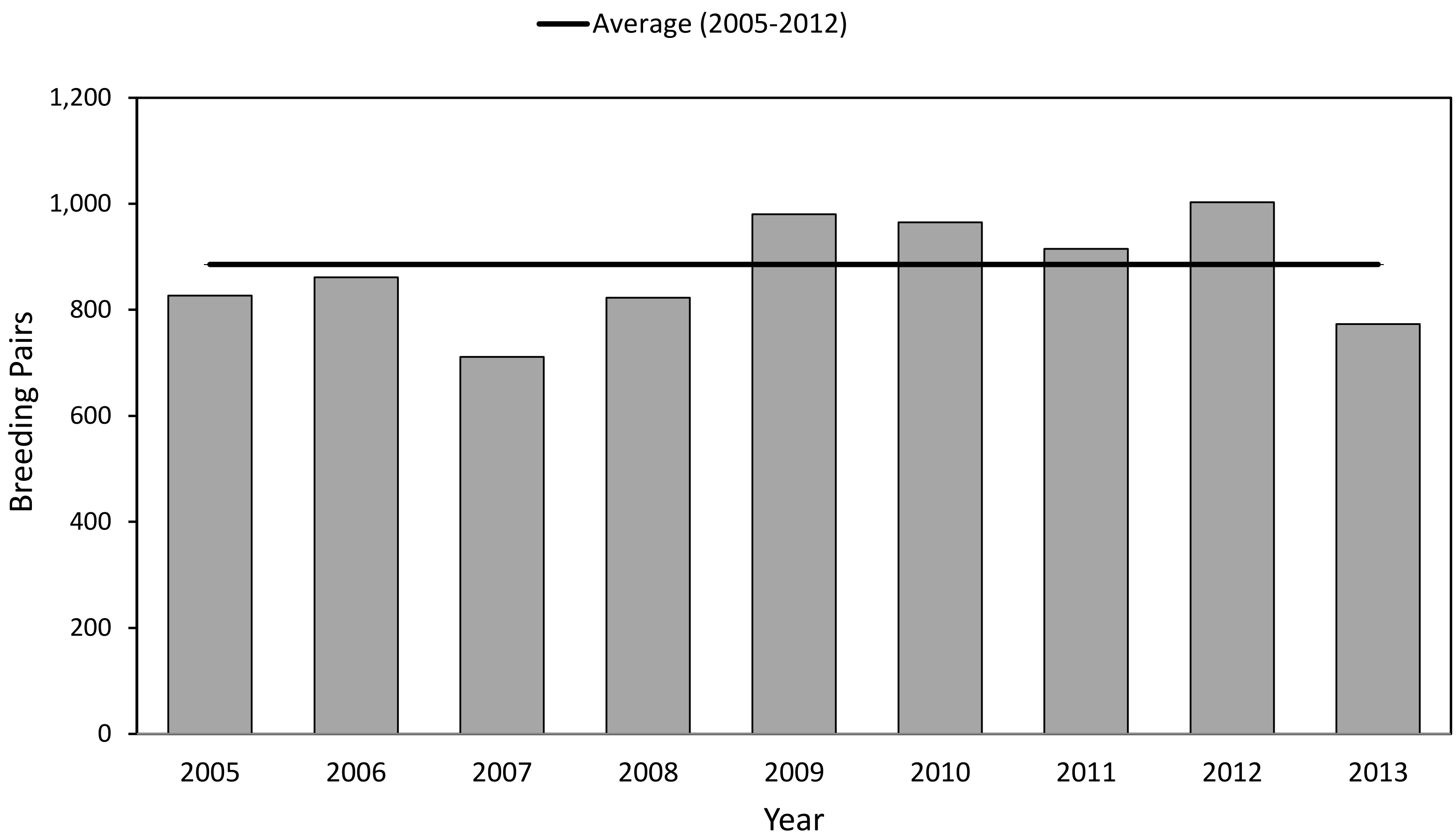


Figure 15. Total numbers of Caspian tern breeding pairs nesting at all known colonies in the Columbia Plateau region during 2005-2013.

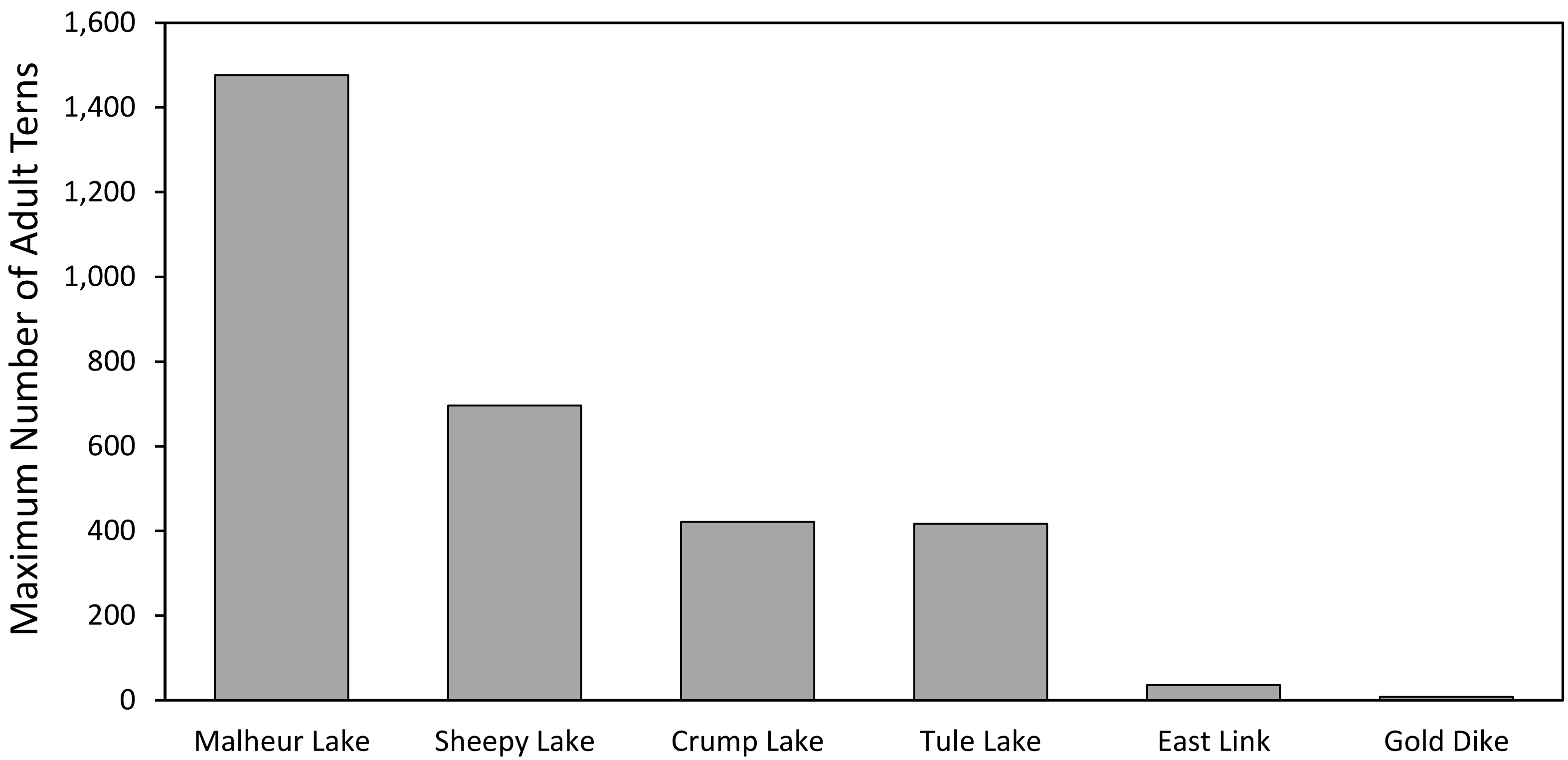


Figure 16. Maximum number of adult Caspian terns counted during the 2013 nesting season on Corps-constructed tern islands in interior Oregon and northeastern California. The Corps-constructed tern island at Fern Ridge Reservoir was not monitored in 2013.

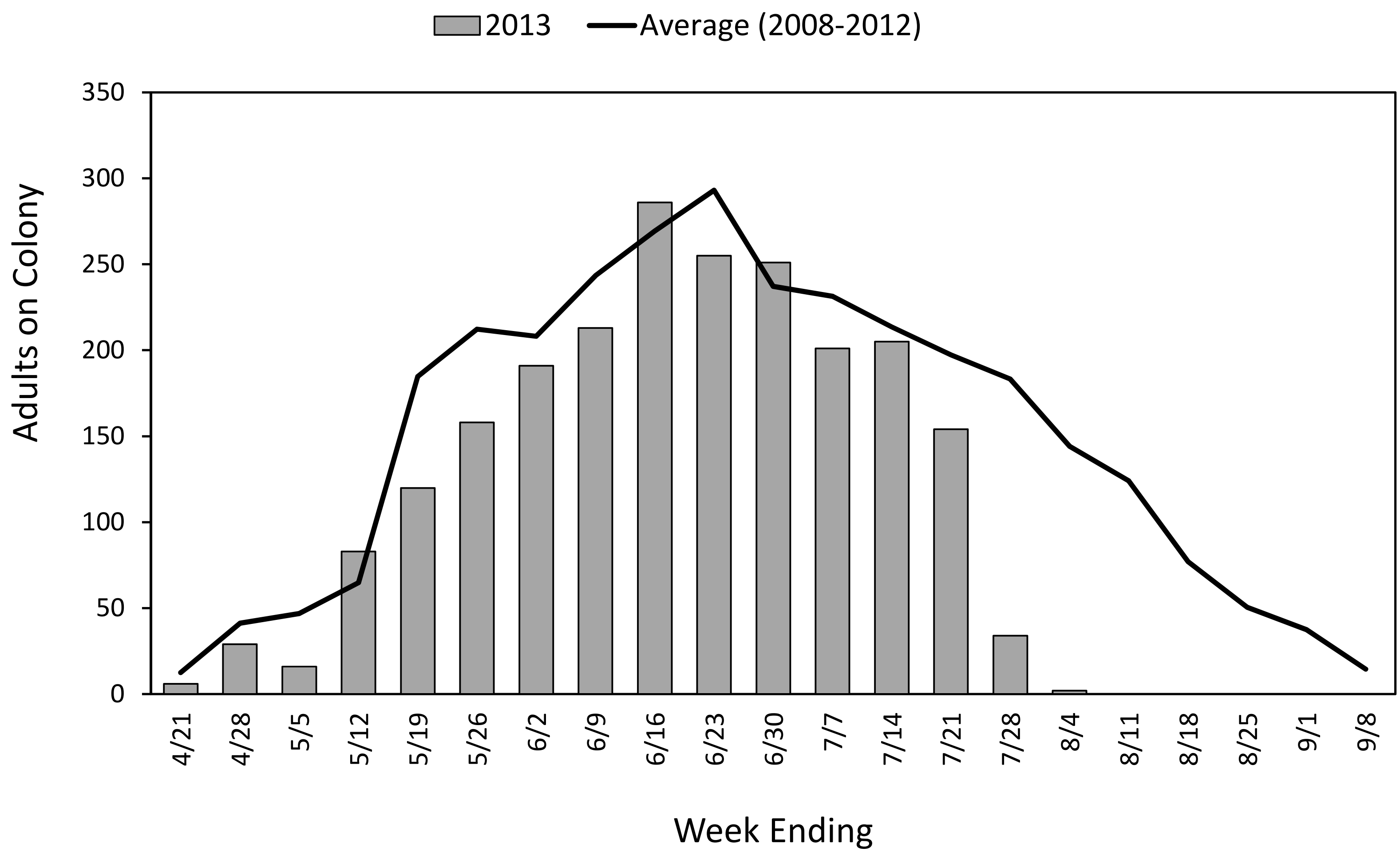


Figure 17. Estimates from the ground of the number of adult Caspian terns on the Corps-constructed tern island at Crump Lake in the Warner Valley, Oregon, by week during the 2013 breeding season.

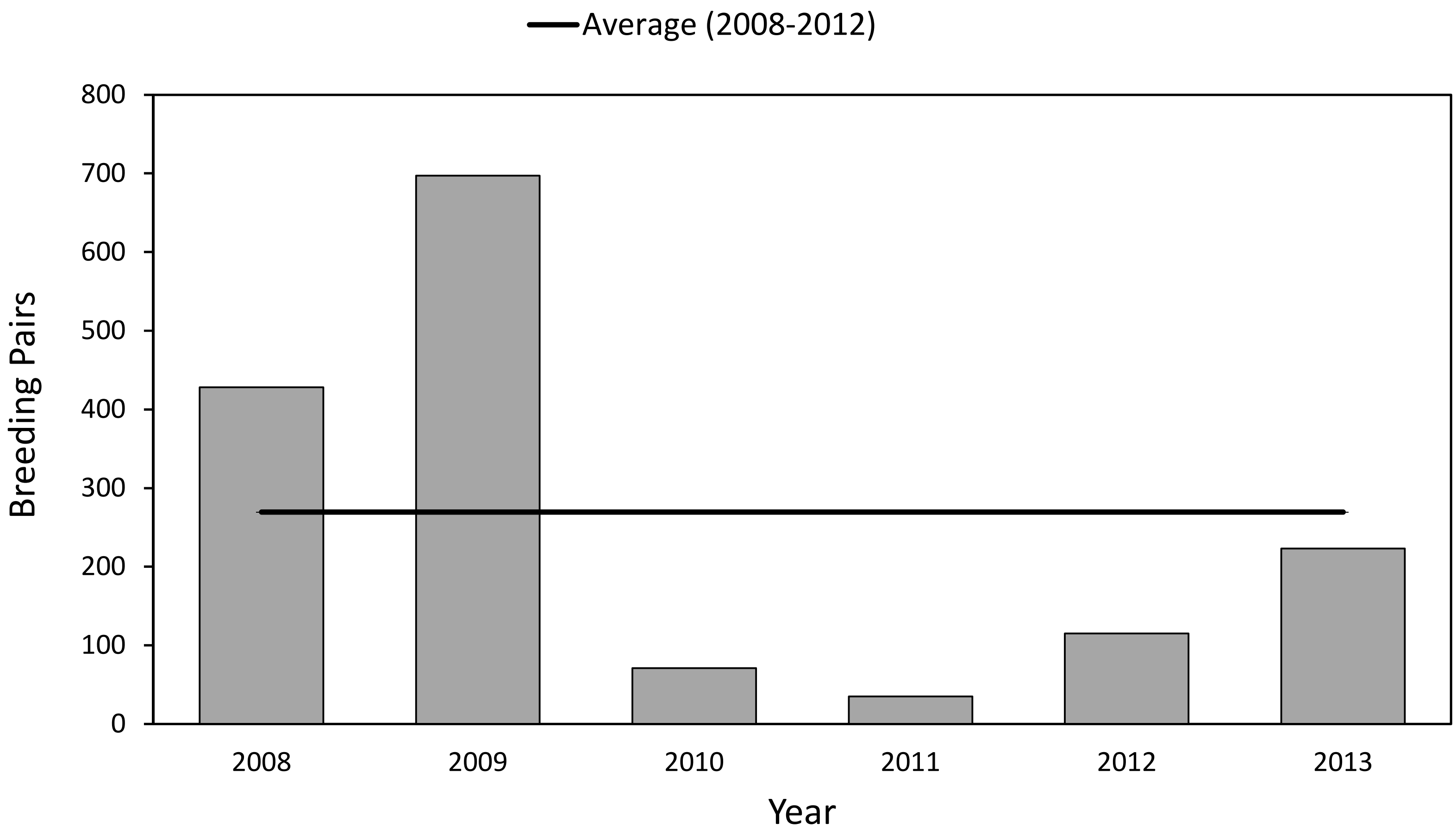


Figure 18. Size of the Caspian tern breeding colony (number of breeding pairs) on the Corps-constructed tern island at Crump Lake in the Warner Valley, Oregon, during the 2008-2013 breeding seasons.

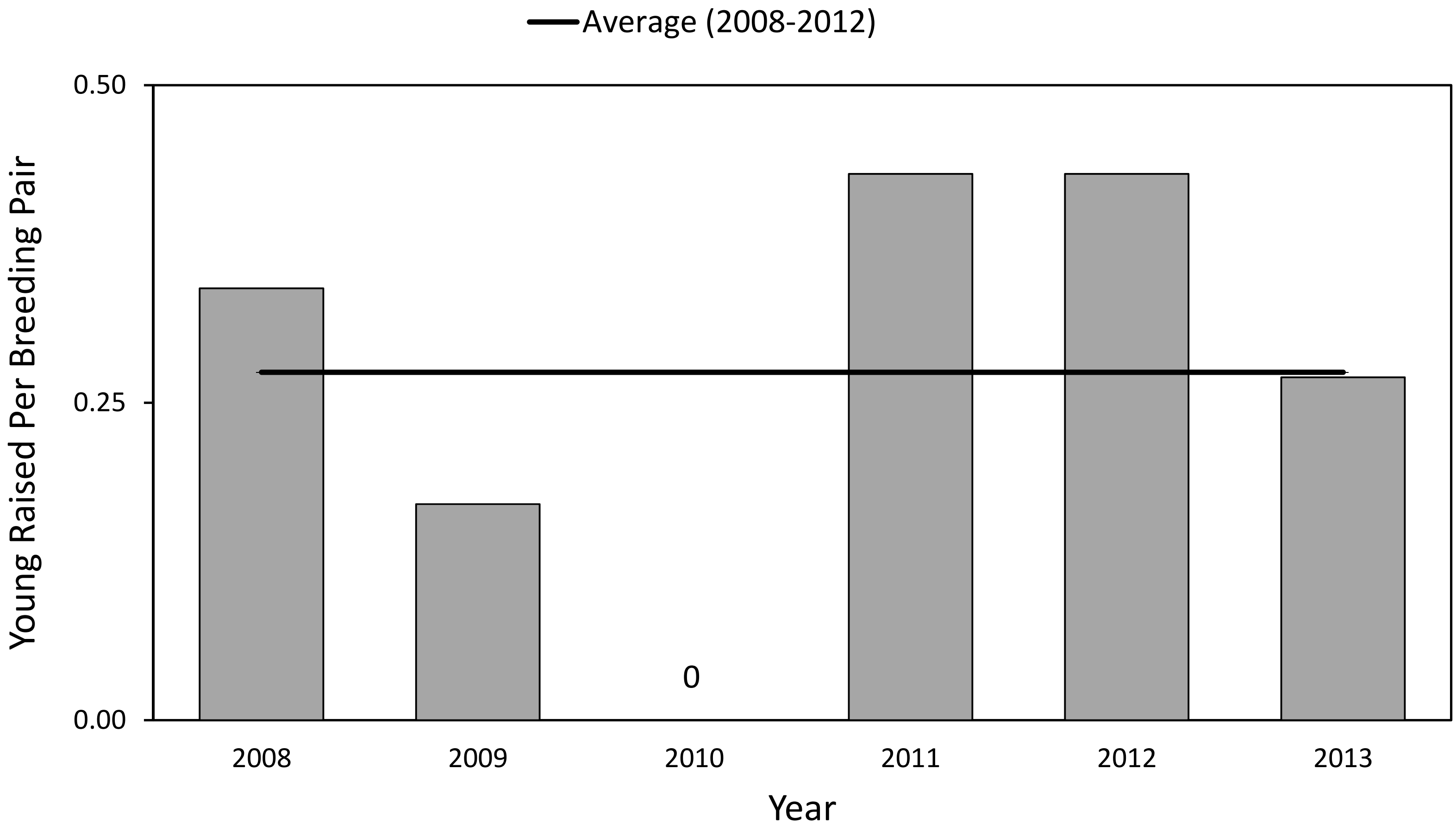


Figure 19. Caspian tern nesting success (average number of young raised per breeding pair) at the Corps-constructed tern island at Crump Lake in the Warner Valley, Oregon, during the 2008-2013 breeding seasons. Caspian terns failed to raise any young at the colony in 2010.

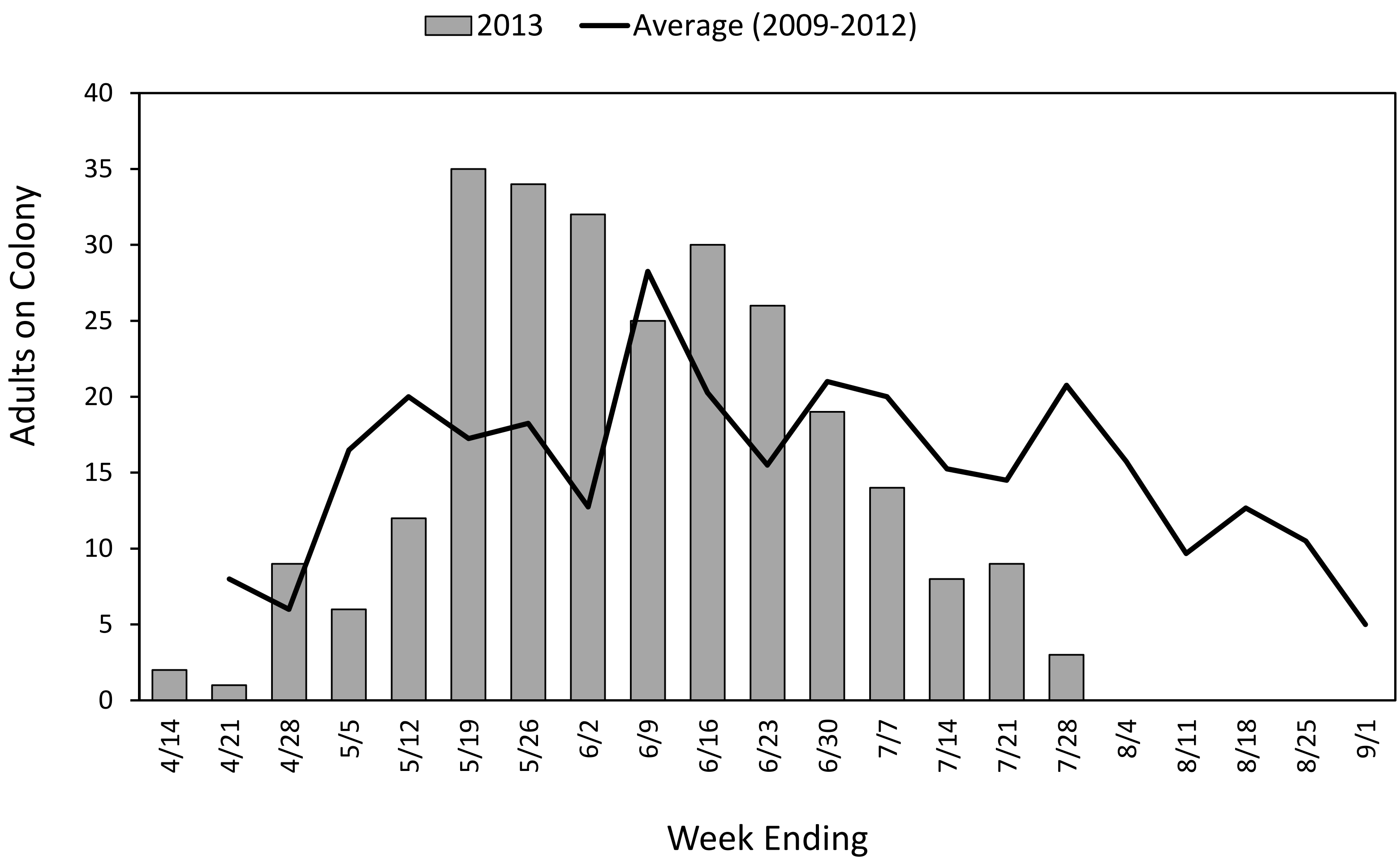


Figure 20. Estimates from the ground of the total number of adult Caspian terns on the Corps-constructed islands in East Link Impoundment and Gold Dike Impoundment at Summer Lake Wildlife Area, Oregon, by week during the 2013 breeding season.

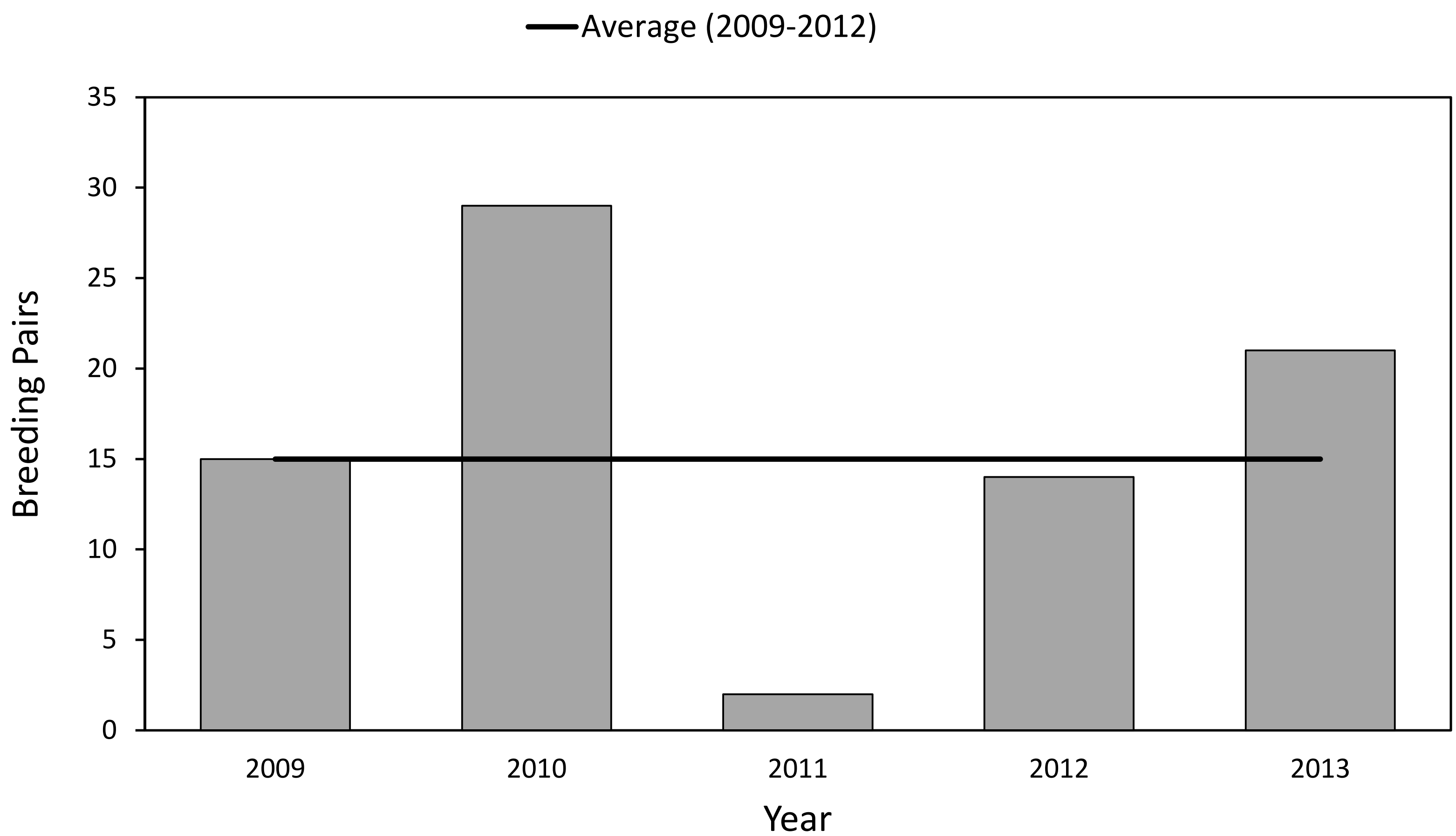


Figure 21. Total size of Caspian tern breeding colonies (number of breeding pairs) on Corps-constructed tern islands in East Link Impoundment, Gold Dike Impoundment, and Dutchy Lake at Summer Lake Wildlife Area, Oregon, during the 2009-2013 breeding seasons. Caspian terns did not nest on the Dutchy Lake tern island during 2010-2012, and the island was removed prior to the 2013 nesting season..

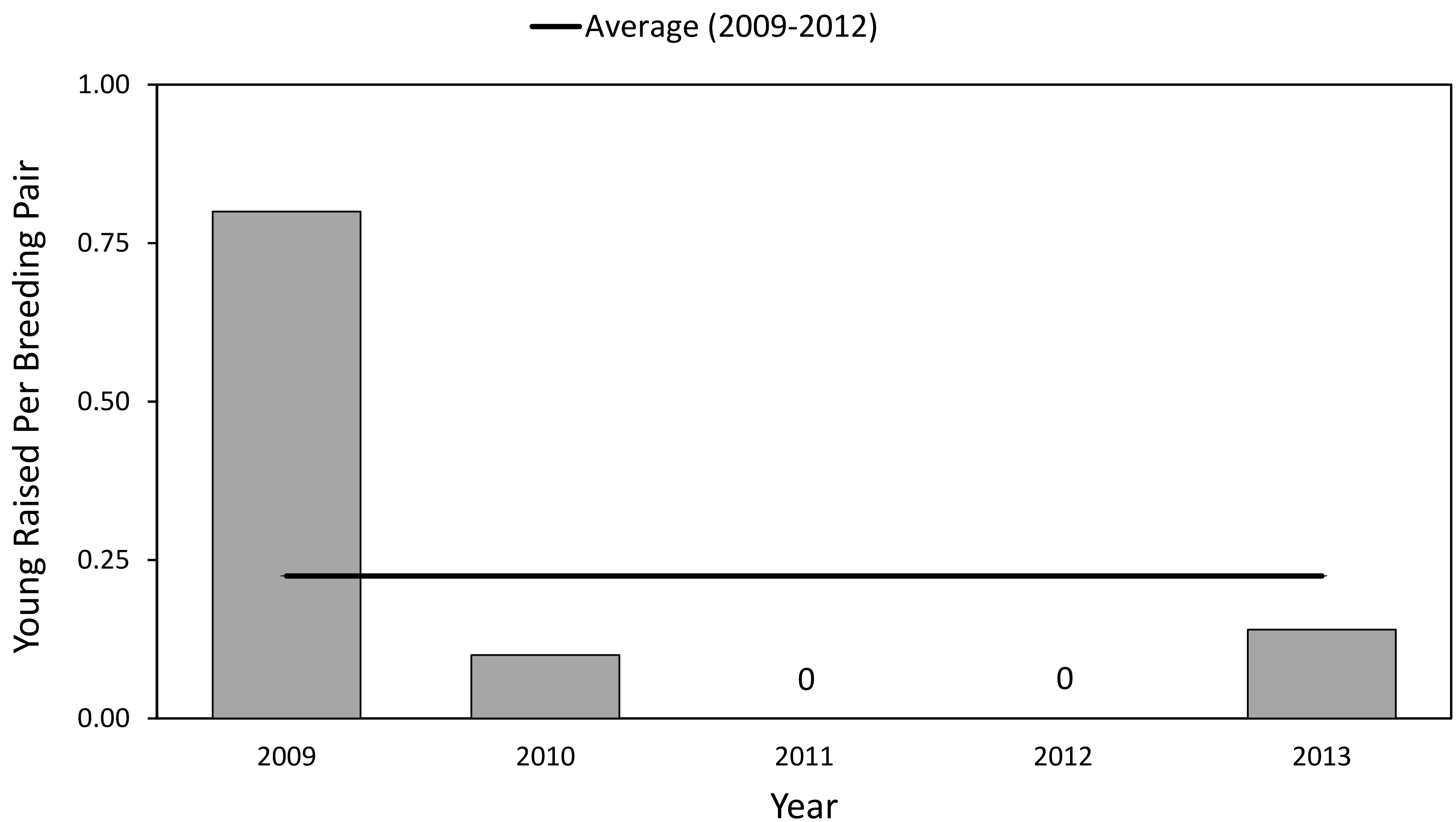


Figure 22. Caspian tern nesting success (average number of young raised per breeding pair) at Corps-constructed tern islands in Summer Lake Wildlife Area (i.e., tern islands in East Link Impoundment, Gold Dike Impoundment, and Dutchy Lake), Oregon, during 2009-2013. Caspian terns did not nest on the Dutchy Lake tern island during 2010-2012, and the island was removed prior to the 2013 nesting season. No young terns were fledged from the East Link tern island during 2011-2012 or the Gold Dike tern island during 2012-2013.

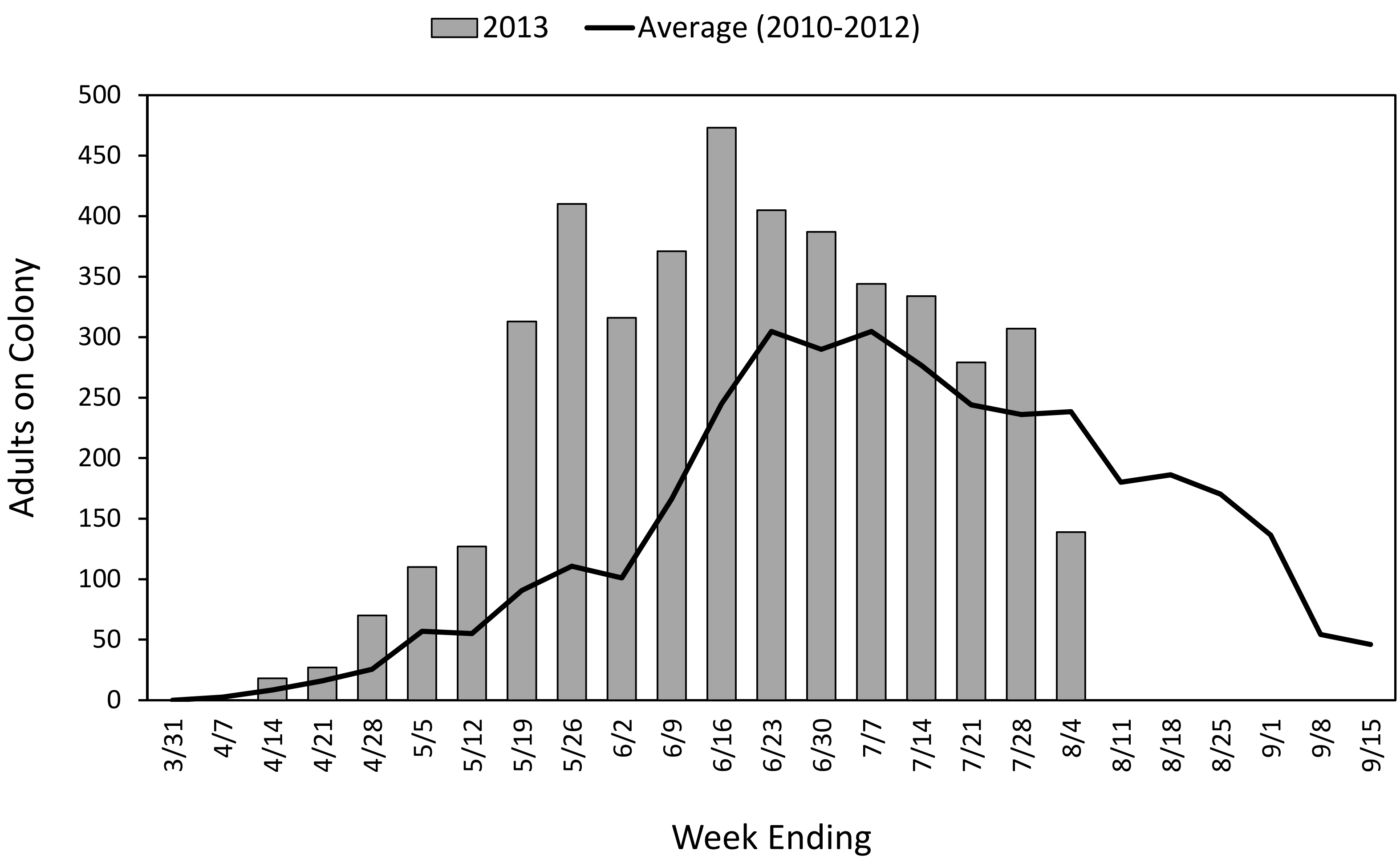


Figure 23. Estimates from the ground of the number of adult Caspian terns on the Corps-constructed tern island at Sheepy Lake in Lower Klamath National Wildlife Refuge, California, by week during the 2013 breeding season.

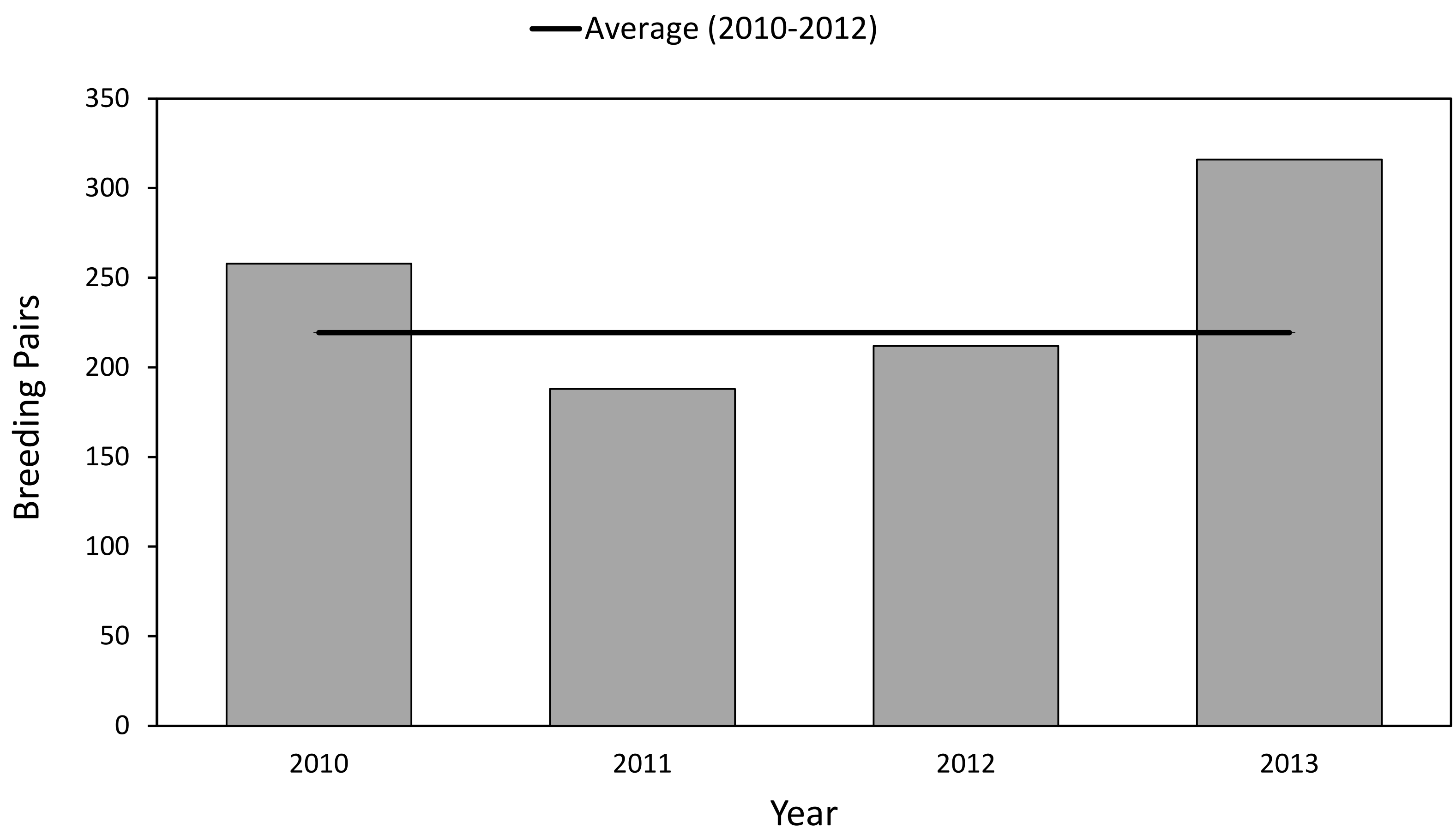


Figure 24. Size of the Caspian tern breeding colony (number of breeding pairs) on the Corps-constructed tern island at Sheepy Lake in Lower Klamath National Wildlife Refuge, California, during the 2010-2013 breeding seasons.

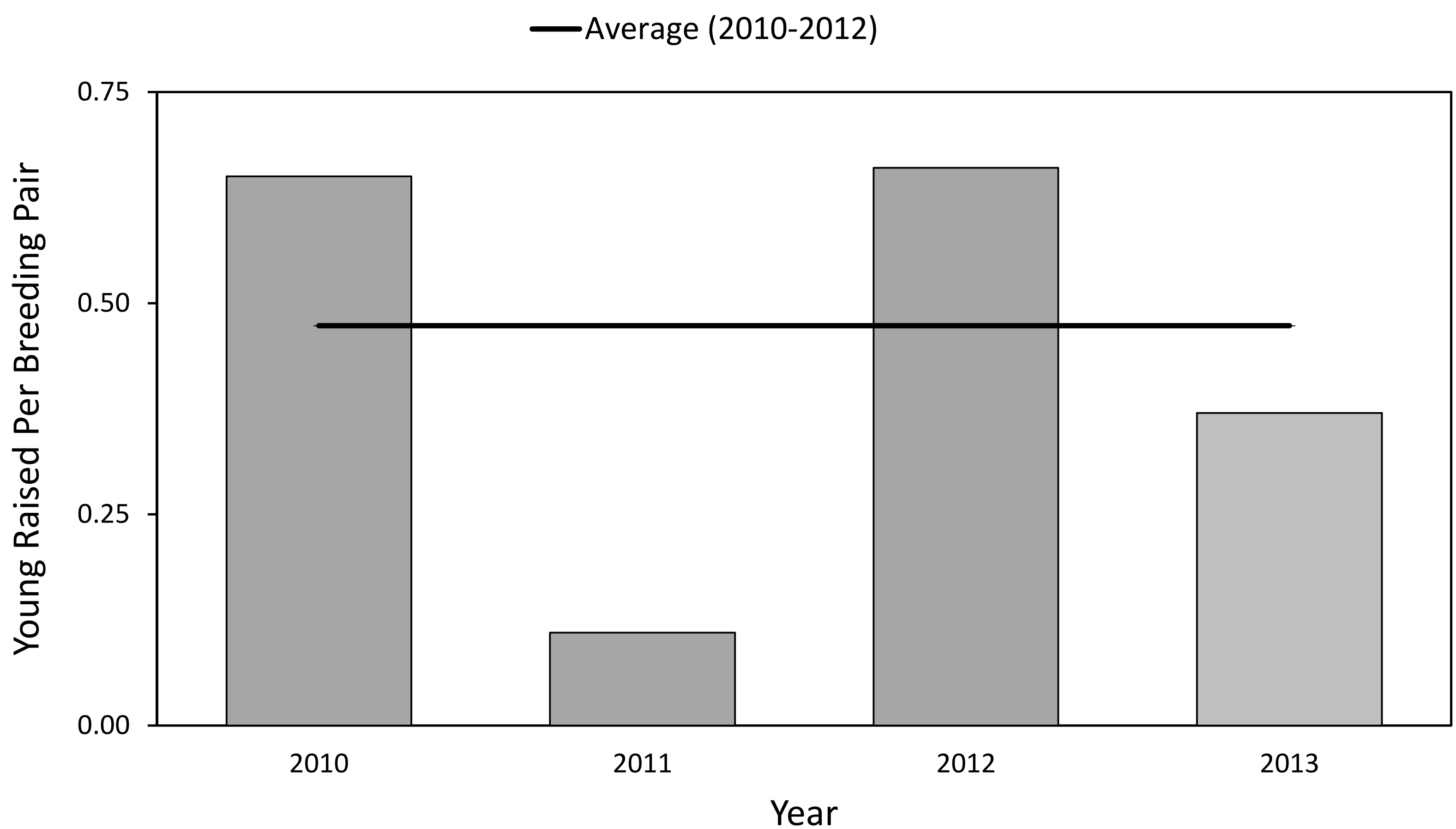


Figure 25. Caspian tern nesting success (average number of young raised per breeding pair) on the Corps-constructed tern island at Sheepy Lake in Lower Klamath National Wildlife Refuge, California, during 2010-2013.

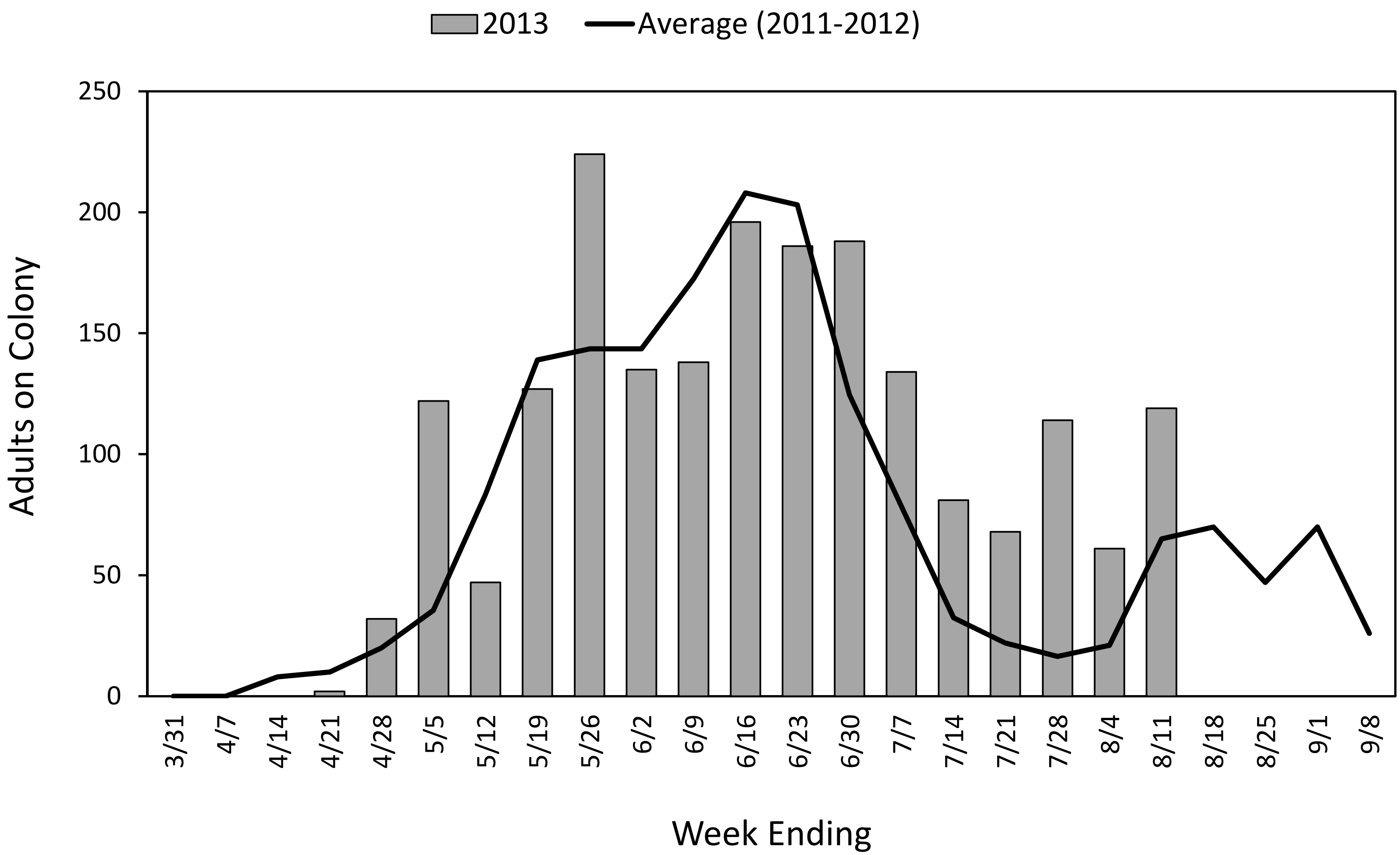


Figure 26. Estimates from the ground of the number of adult Caspian terns on the Corps-constructed tern island at Tule Lake Sump 1B in Tule Lake National Wildlife Refuge, California, by week during the 2013 breeding season.

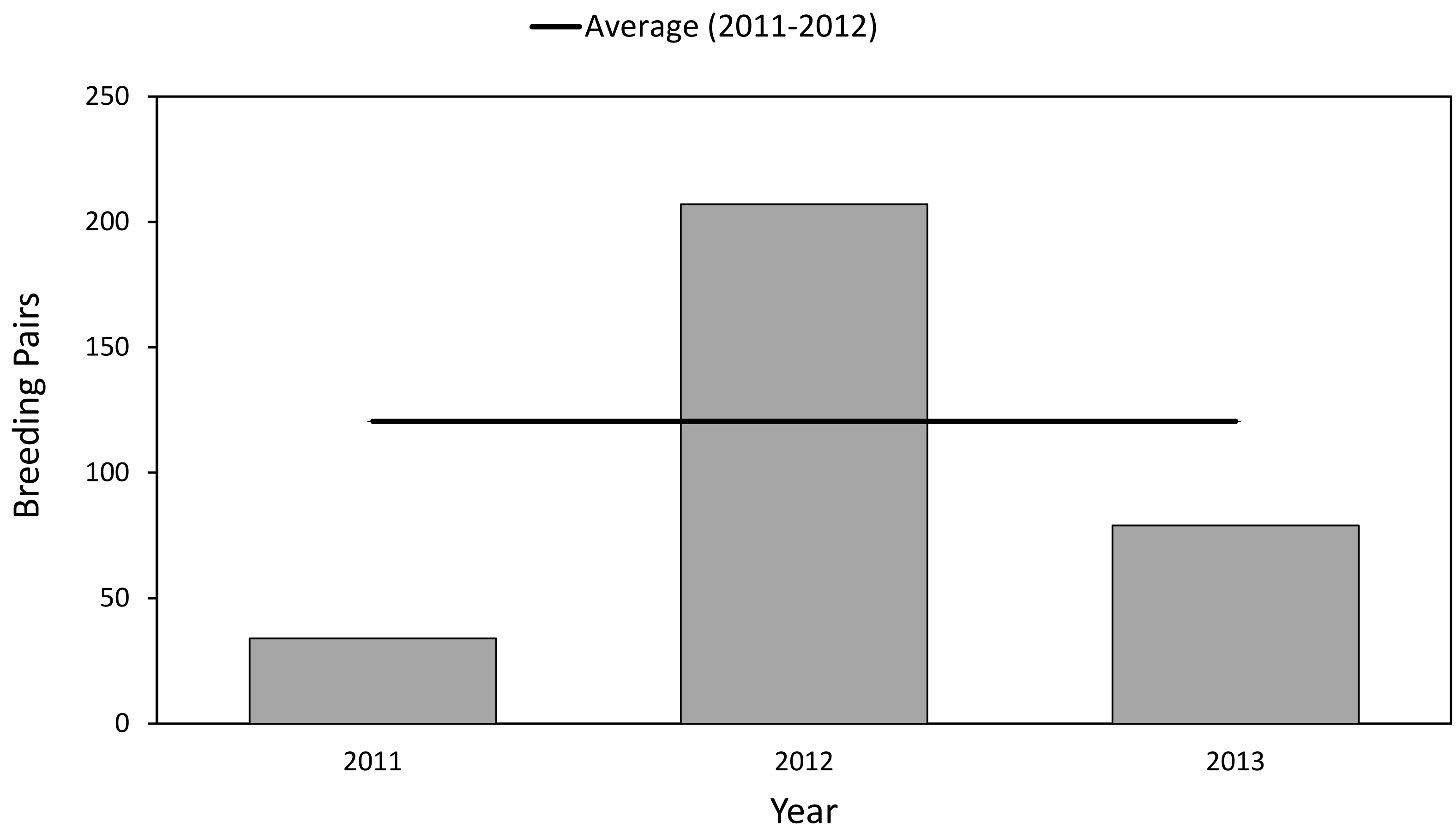


Figure 27. Size of the Caspian tern breeding colony (number of breeding pairs) on the Corps-constructed tern island at Tule Lake Sump 1B in Tule Lake National Wildlife Refuge, California, during the 2011-2013 breeding seasons.

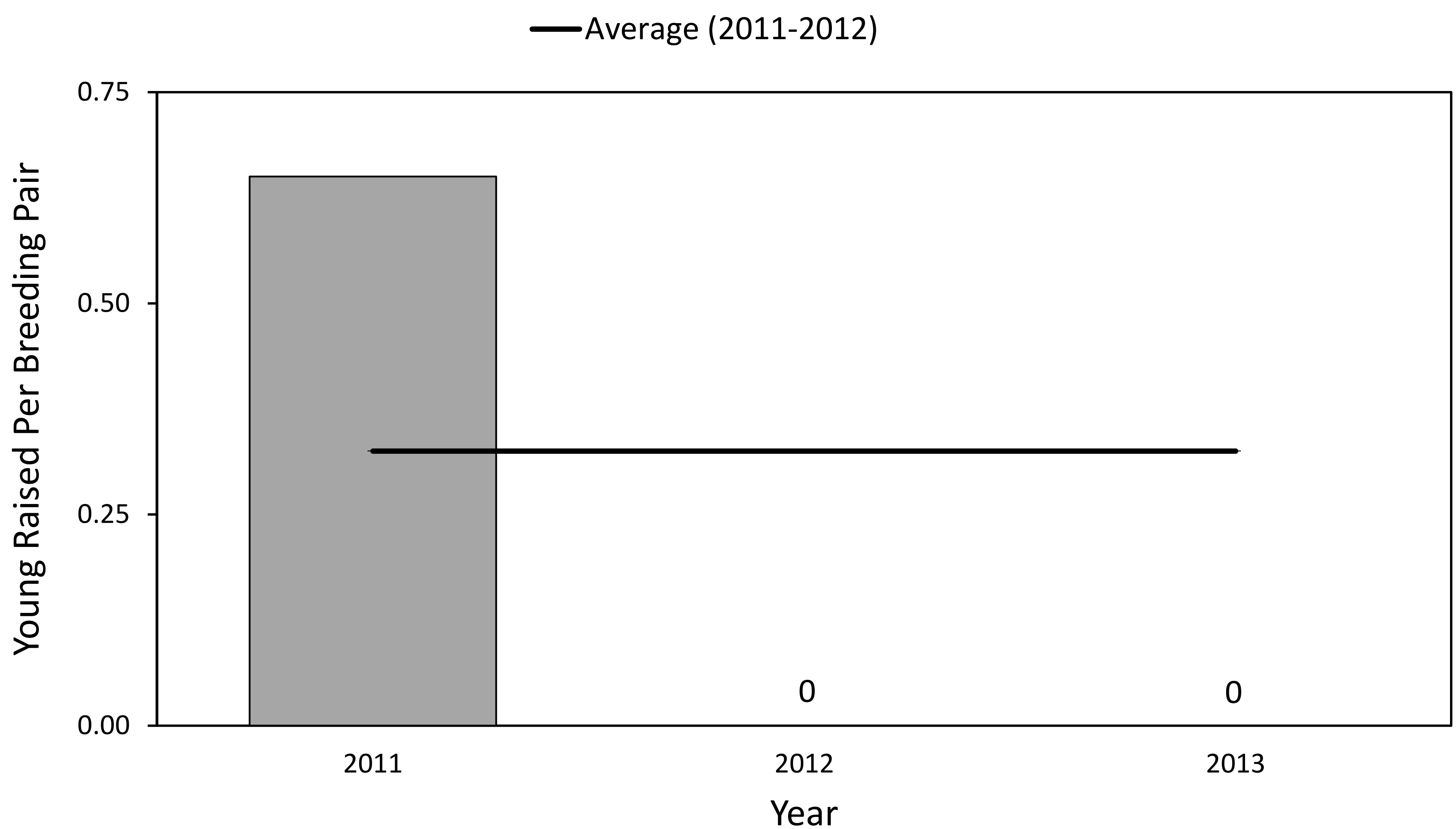


Figure 28. Caspian tern nesting success (average number of young raised per breeding pair) on the Corps-constructed tern island at Tule Lake Sump 1B in Tule Lake National Wildlife Refuge, California, during 2011-2013.

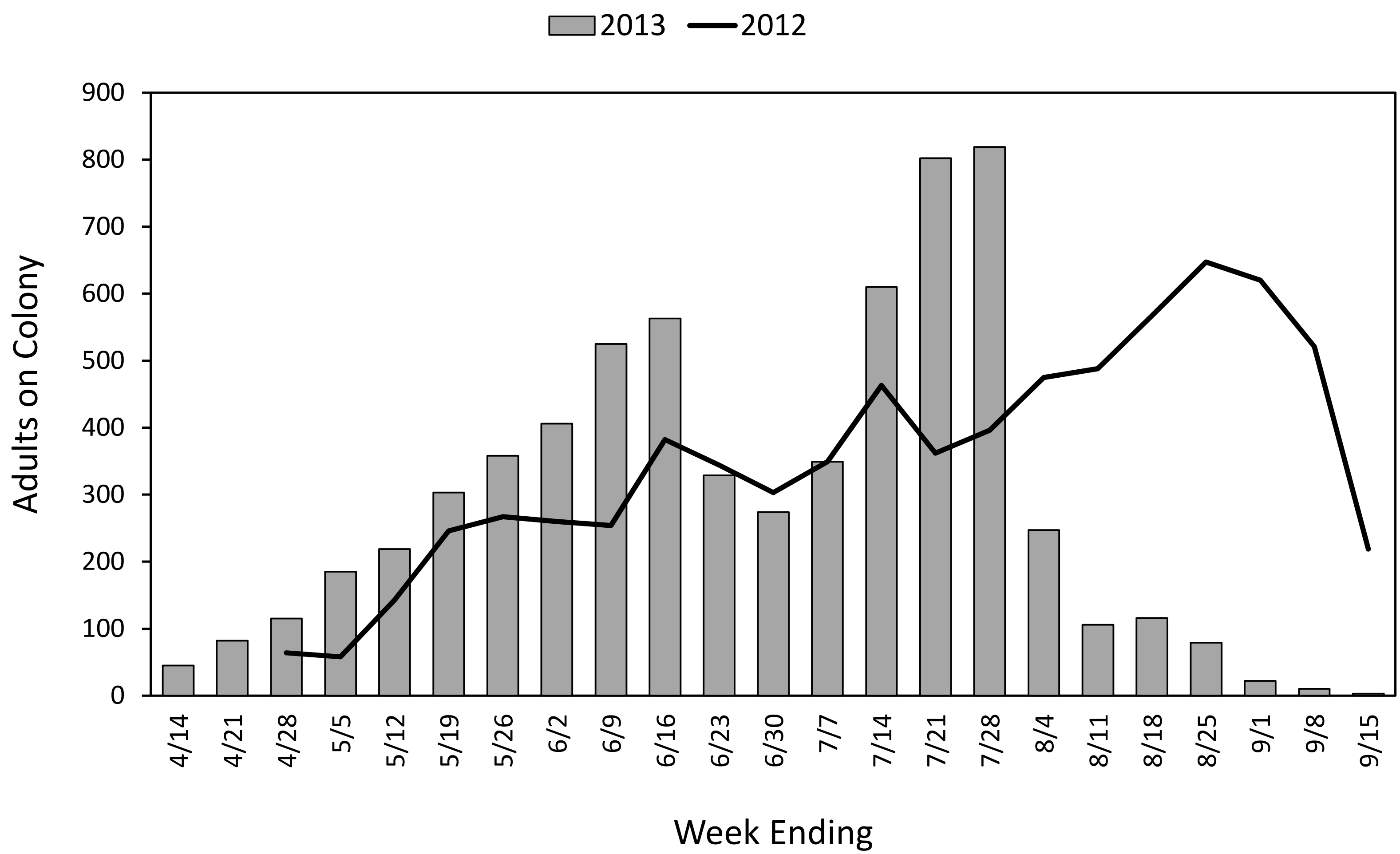


Figure 29. Estimates from the ground of the number of adult Caspian terns on the Corps-constructed tern island at Malheur Lake in Malheur National Wildlife Refuge, Oregon, by week during the 2013 breeding season.

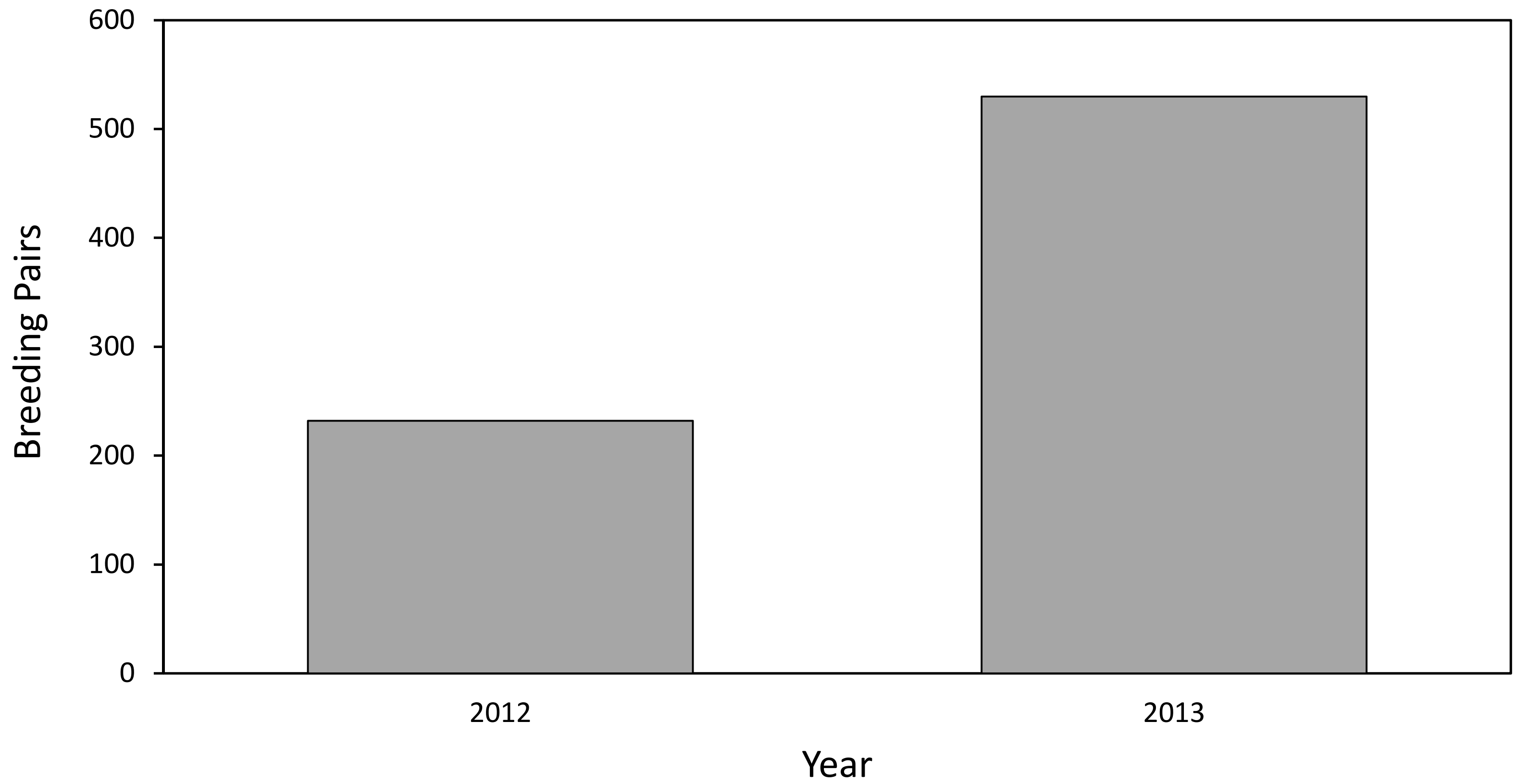


Figure 30. Size of the Caspian tern breeding colony (number of breeding pairs) on the Corps-constructed tern island at Malheur Lake in Malheur National Wildlife Refuge, Oregon, during the 2012-2013 breeding seasons.

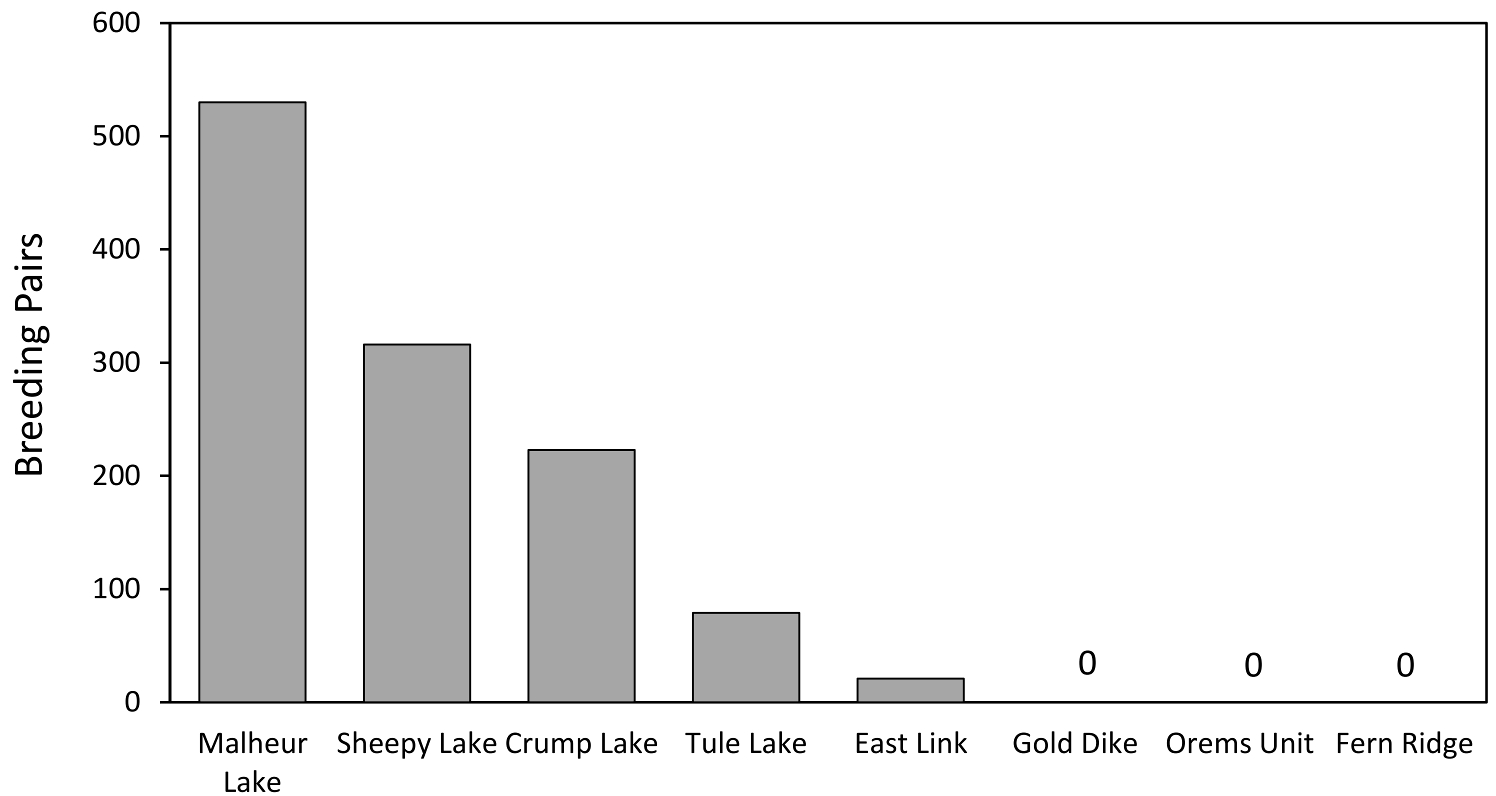


Figure 31. Sizes of Caspian tern breeding colonies (numbers of breeding pairs) on Corps-constructed tern islands in interior Oregon and northeastern California during the 2013 breeding season.

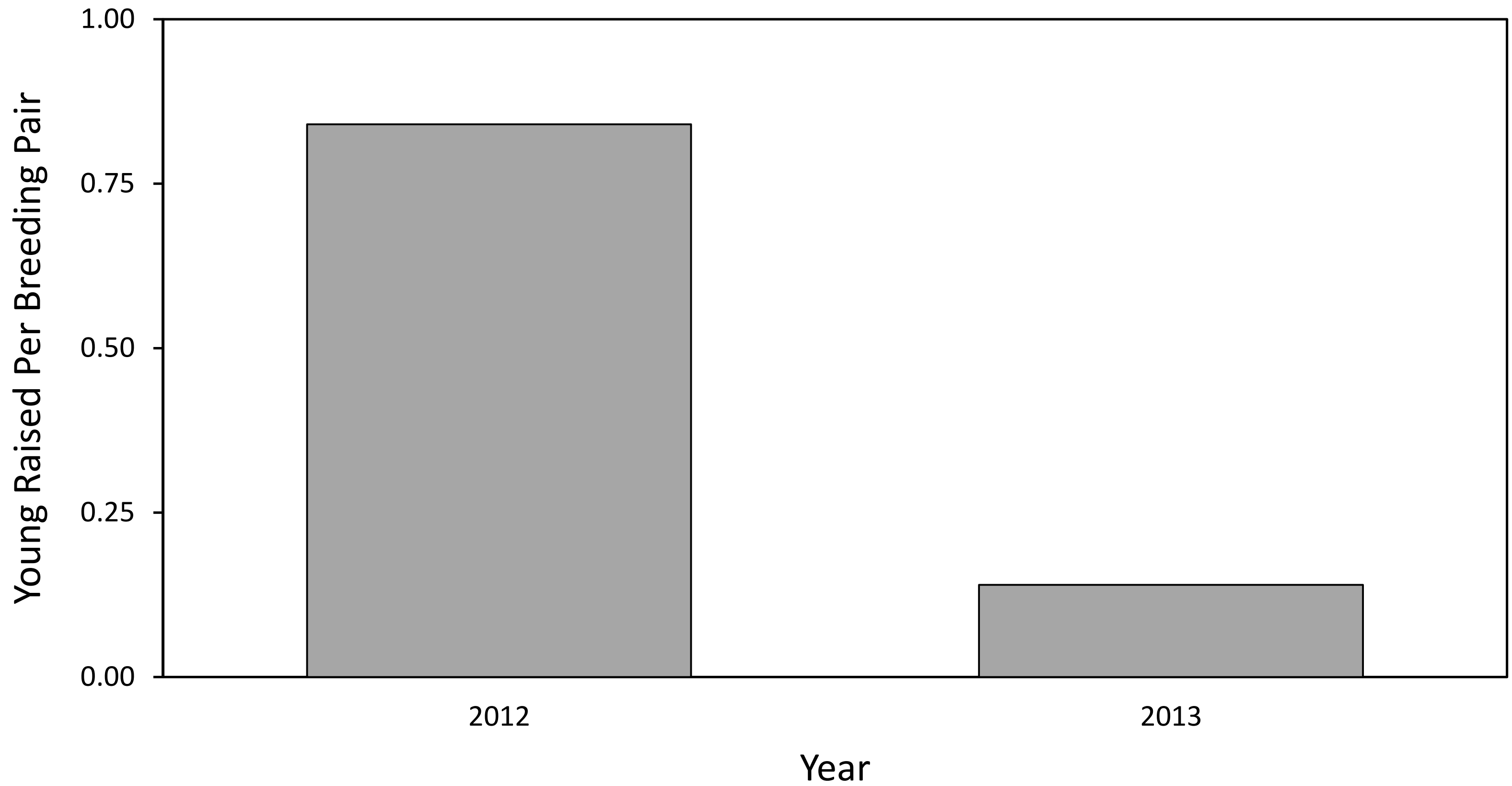


Figure 32. Caspian tern nesting success (average number of young raised per breeding pair) on the Corps-constructed tern island at Malheur Lake in Malheur National Wildlife Refuge, Oregon, during 2012-2013.

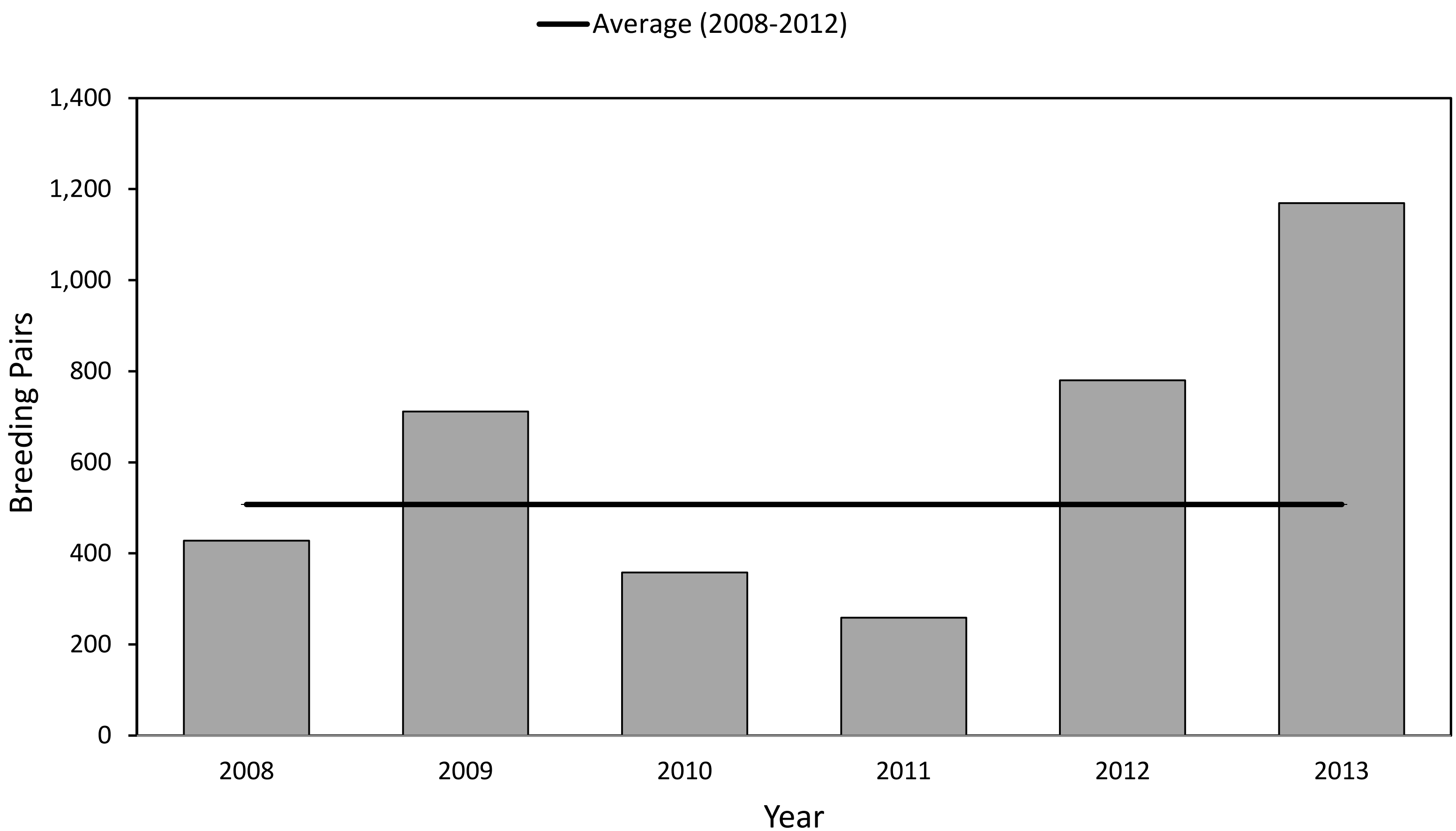


Figure 33. Total numbers of Caspian tern breeding pairs nesting on Corps-constructed tern islands in interior Oregon and northeastern California during 2008-2013.

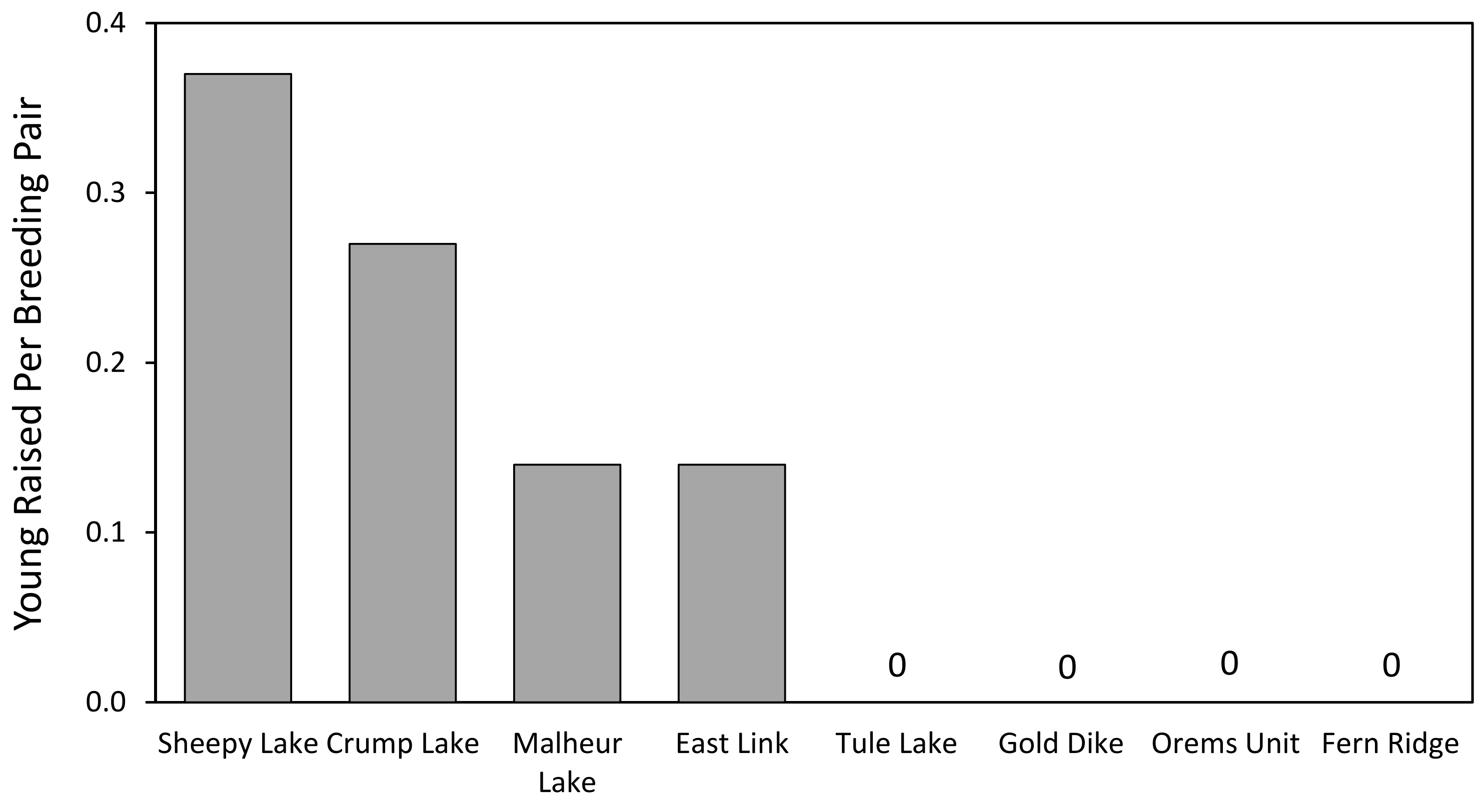


Figure 34. Caspian tern nesting success (average number of young raised per breeding pair) on Corps-constructed tern islands in interior Oregon and northeastern California during the 2013 breeding season.

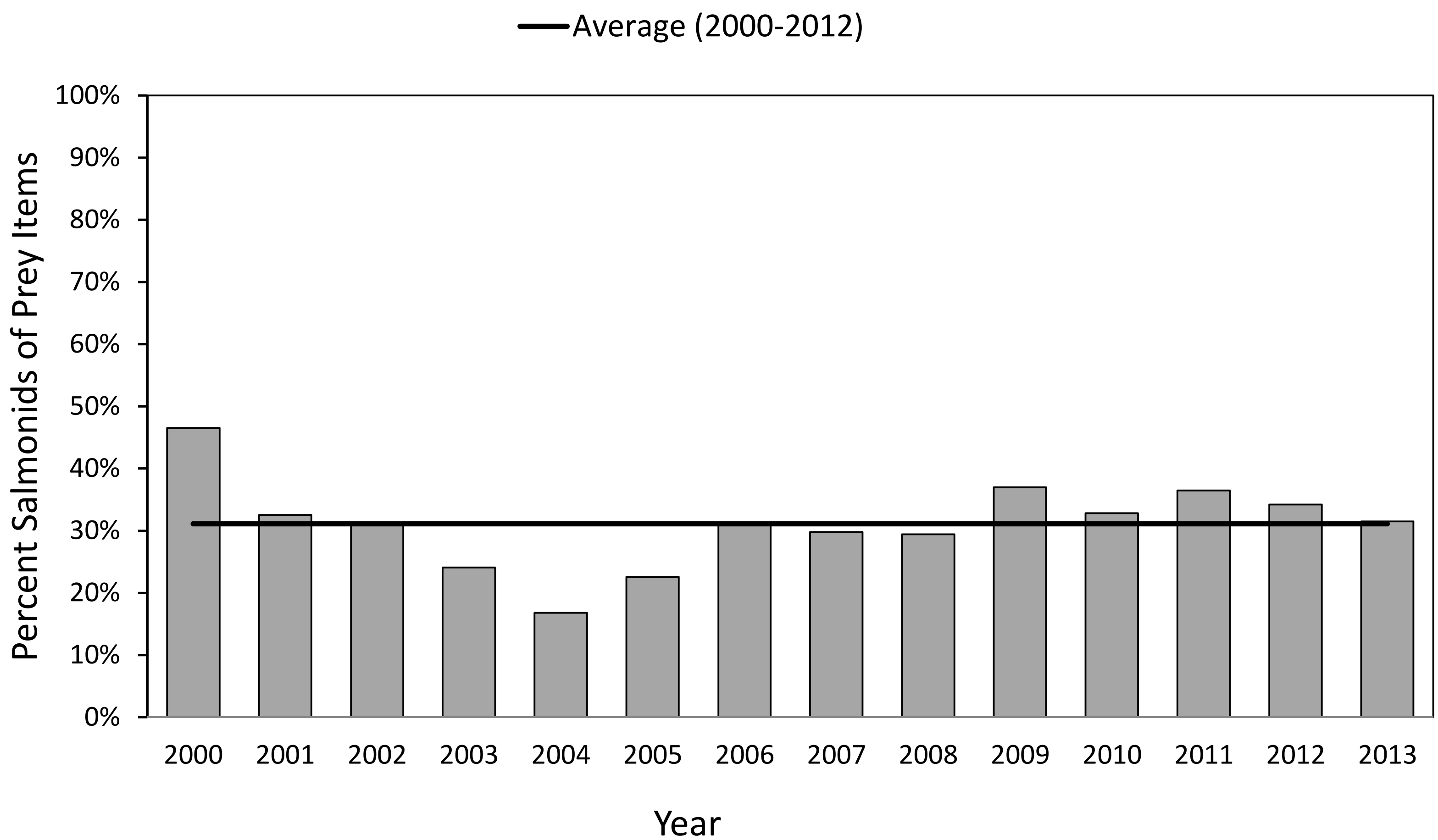


Figure 35. Average annual proportion of juvenile salmonids in the diet (percent of prey items) of Caspian terns nesting on East Sand Island in the Columbia River estuary during the 2000-2013 breeding seasons.

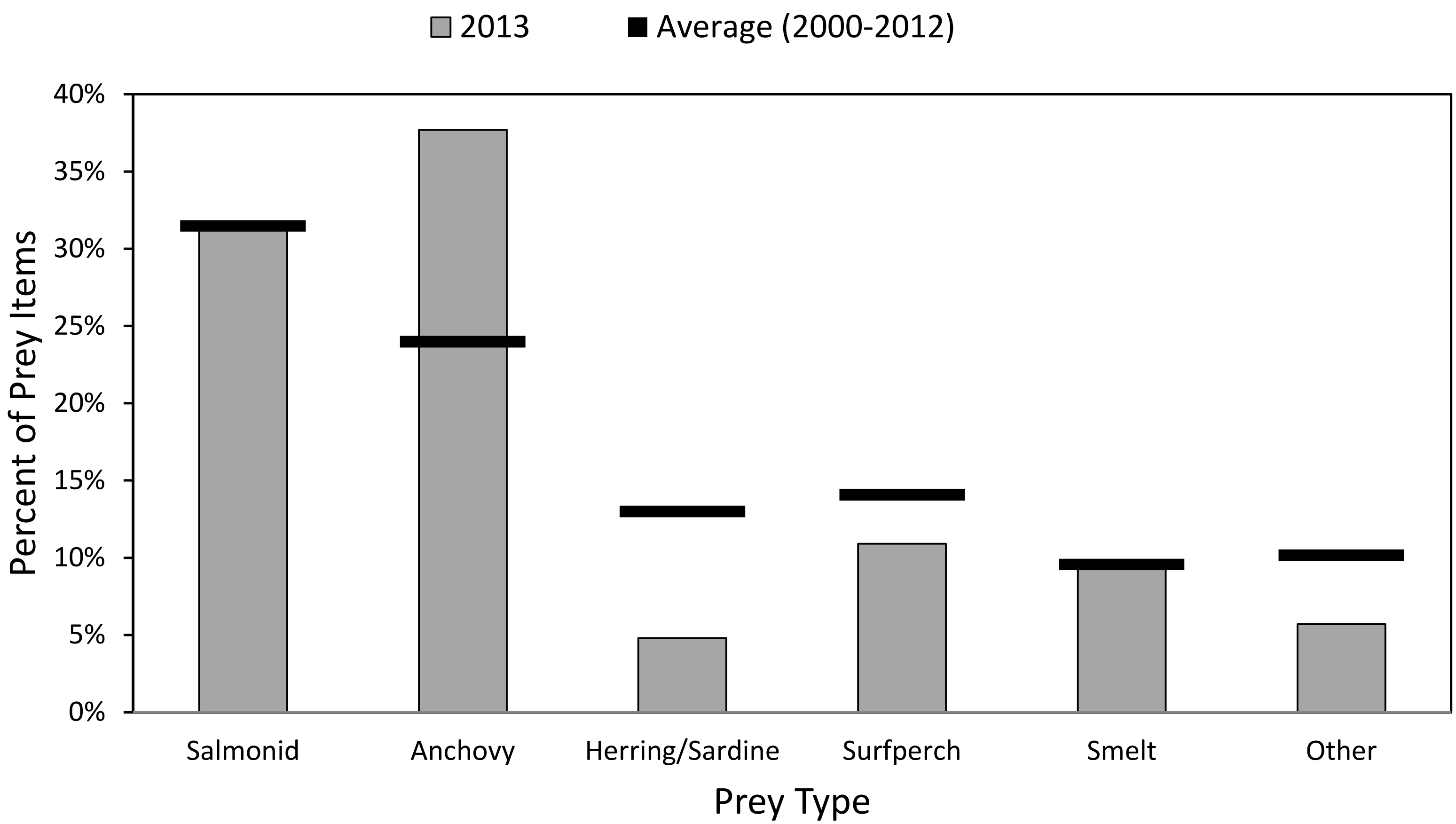


Figure 36. Diet composition (percent of identified prey items) of Caspian terns nesting on East Sand Island in the Columbia River estuary during the 2013 breeding season. Diet composition was based on fish visually identified on-colony in Caspian tern bill-loads.

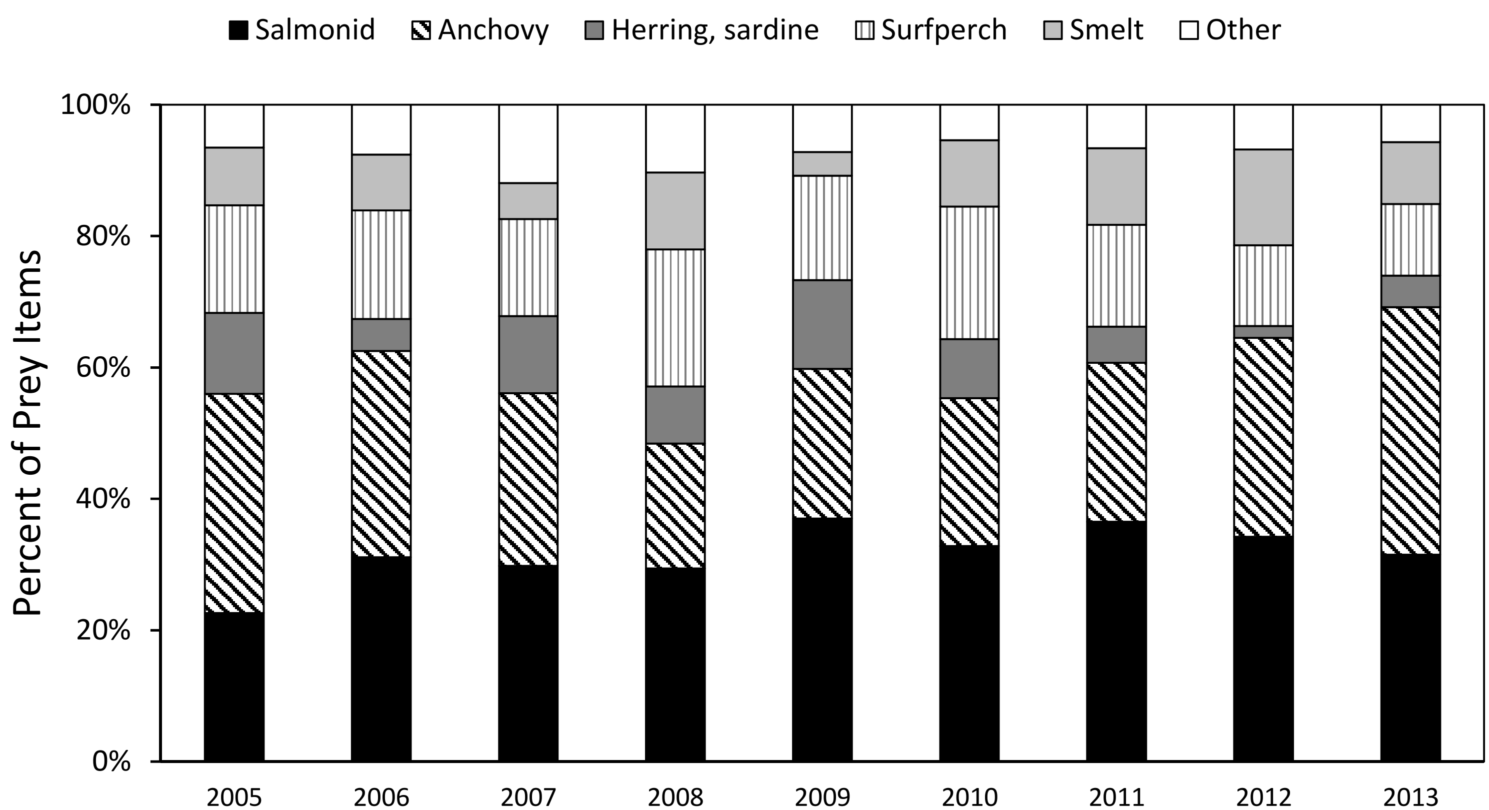


Figure 37. Annual diet composition (percent of prey items) of Caspian terns nesting on East Sand Island in the Columbia River estuary during the 2005-2013 breeding seasons. Diet composition was based on fish visually identified on-colony in Caspian tern bill-loads.

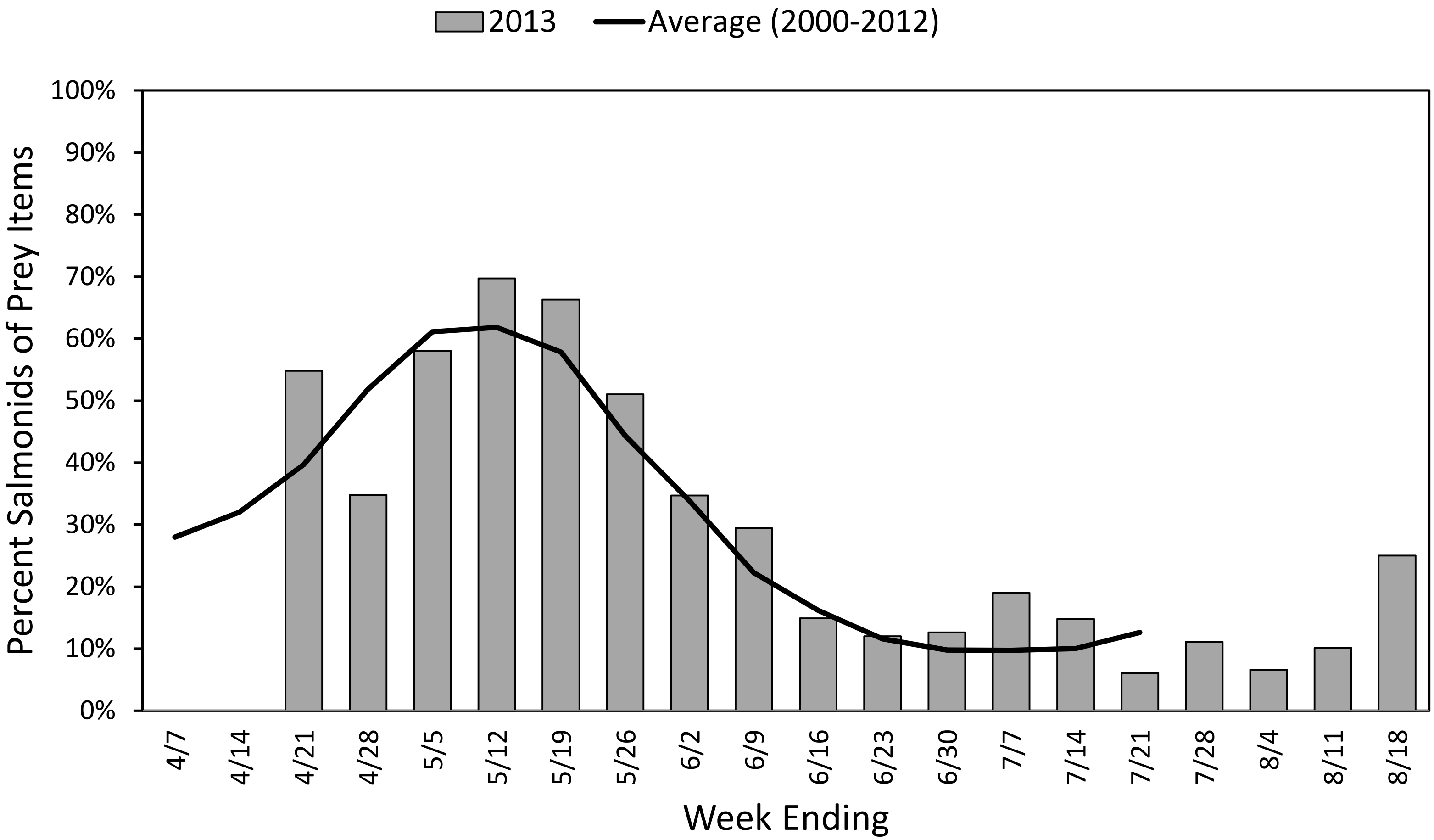
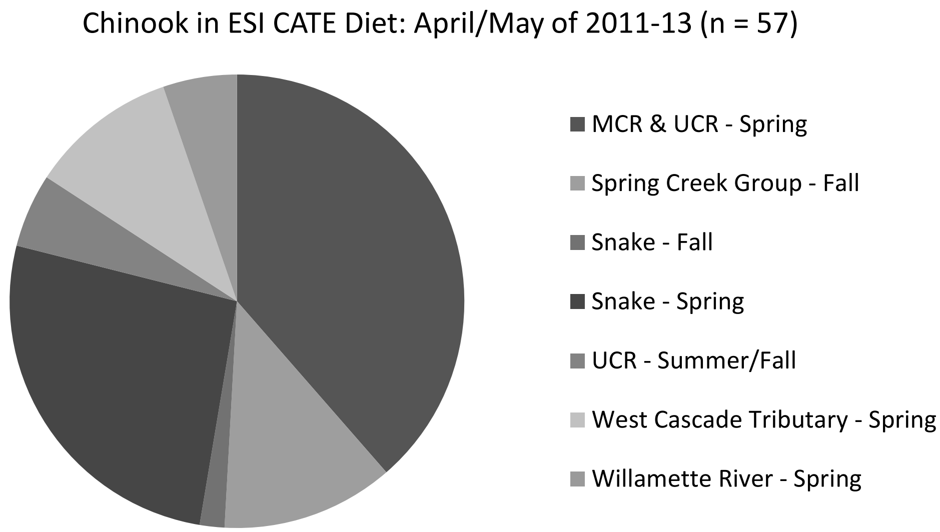


Figure 38. Proportion of juvenile salmonids in the diet (percent of prey items) of Caspian terns nesting on East Sand Island in the Columbia River estuary, by week during the 2013 breeding season.



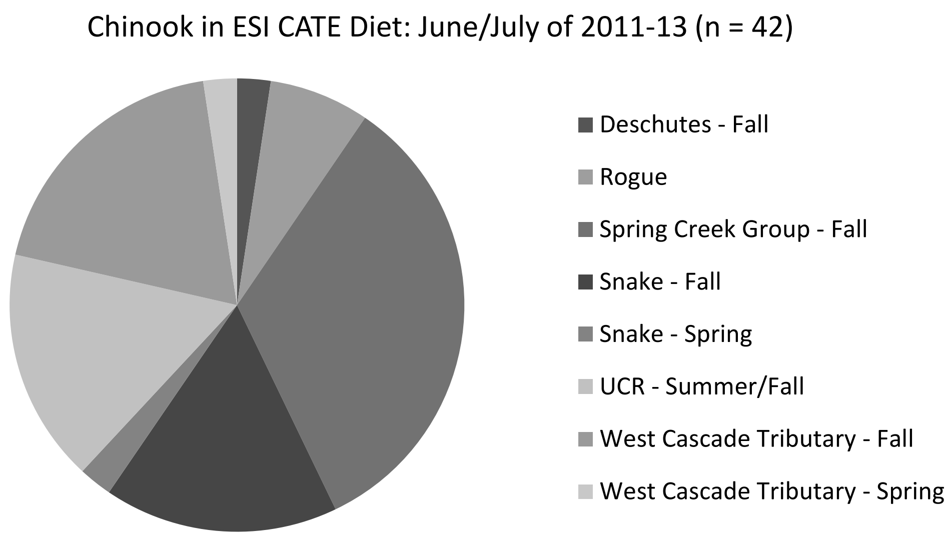
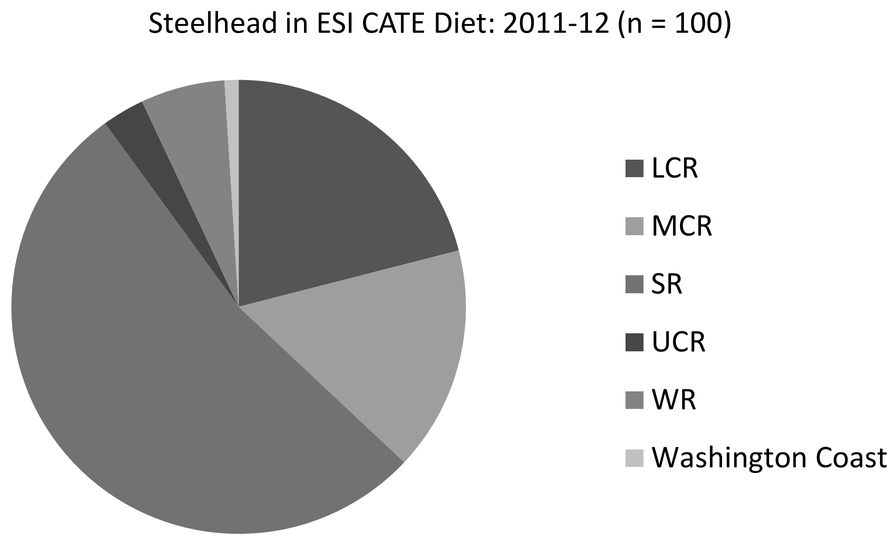


Figure 39. Genetic stock of origin for Chinook salmon in the diet of Caspian terns (CATE) nesting at East Sand Island (ESI) in the Columbia River estuary. Genetic stock identification of salmonids was performed by D. Kuligowski, NOAA Fisheries, on bill-load fish obtained from Caspian terns returning to the East Sand Island colony during the 2011-2013 breeding seasons. The Rogue River fall run stock was introduced to the lower Columbia River as part of a select area fishery enhancement project (North et al. 2006).



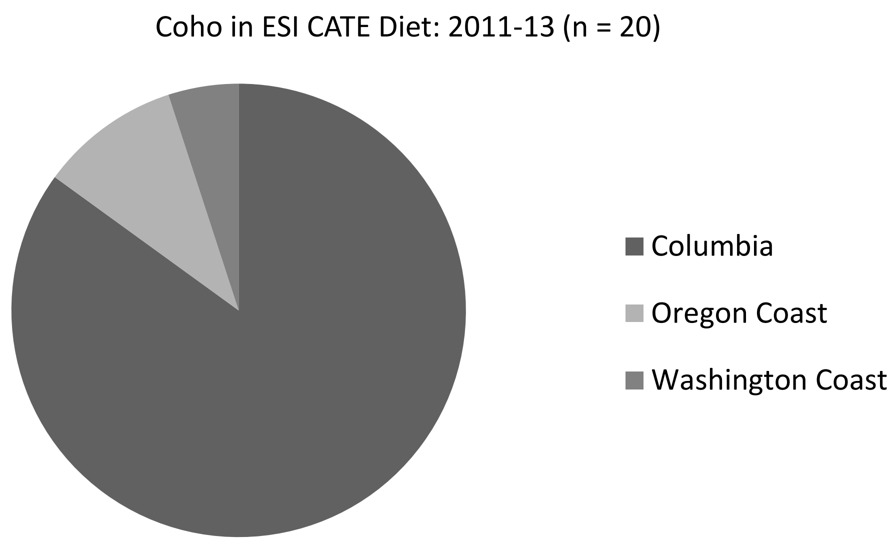


Figure 40. Genetic stock of origin for steelhead trout and coho salmon in the diet of Caspian terns (CATE) nesting on East Sand Island (ESI) in the Columbia River estuary. Genetic stock identification of salmonids was performed by D. Kuligowski, NOAA Fisheries, on bill-load fish obtained from Caspian terns returning to the East Sand Island colony during the 2011-2013 breeding seasons. Only a small sample of coho salmon collected from Caspian terns (n = 20) was submitted for genetic analysis.

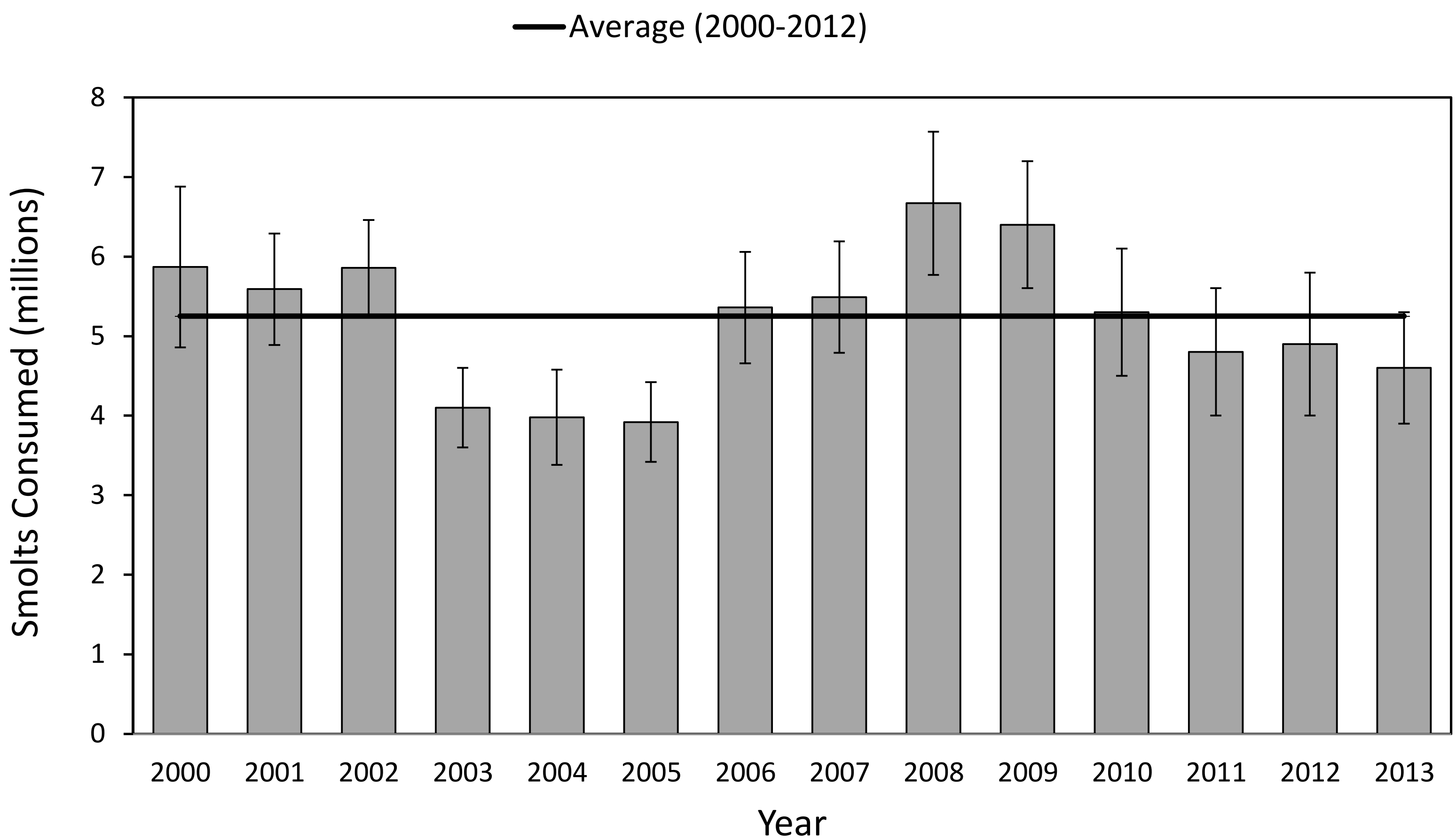


Figure 41. Estimated total annual consumption of juvenile salmonids by Caspian terns nesting on East Sand Island in the Columbia River estuary during the 2000-2013 breeding seasons. Estimates are based on fish identified in tern bill-loads on-colony and bioenergetics calculations. Error bars represent 95% confidence intervals for the number of smolts consumed.

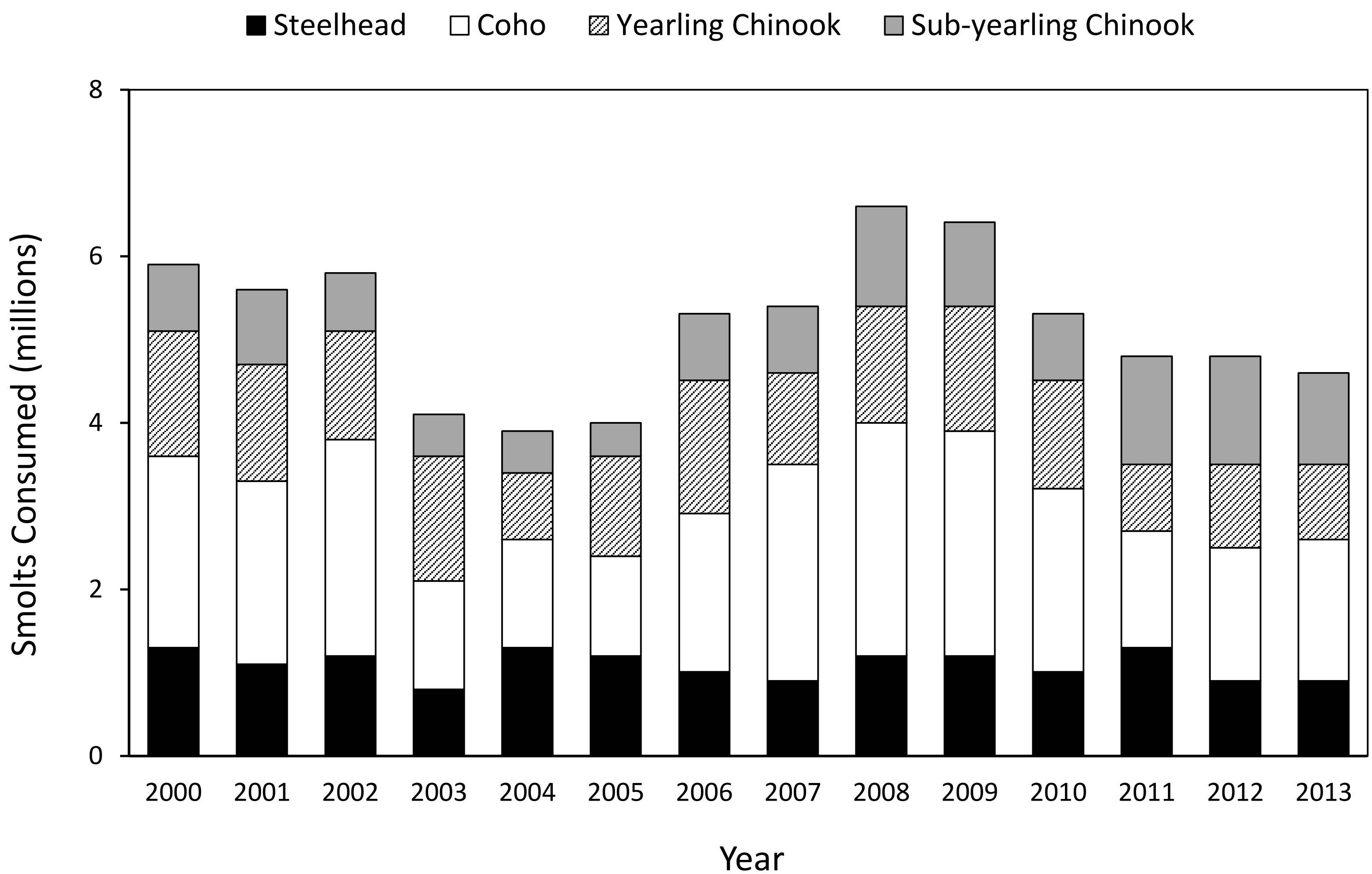


Figure 42. Estimated total annual consumption of four species/run types of juvenile salmonids by Caspian terns nesting on East Sand Island in the Columbia River estuary during the 2000-2013 breeding seasons. Estimates are based on fish collected from tern bill-loads near the colony and bioenergetics calculations.

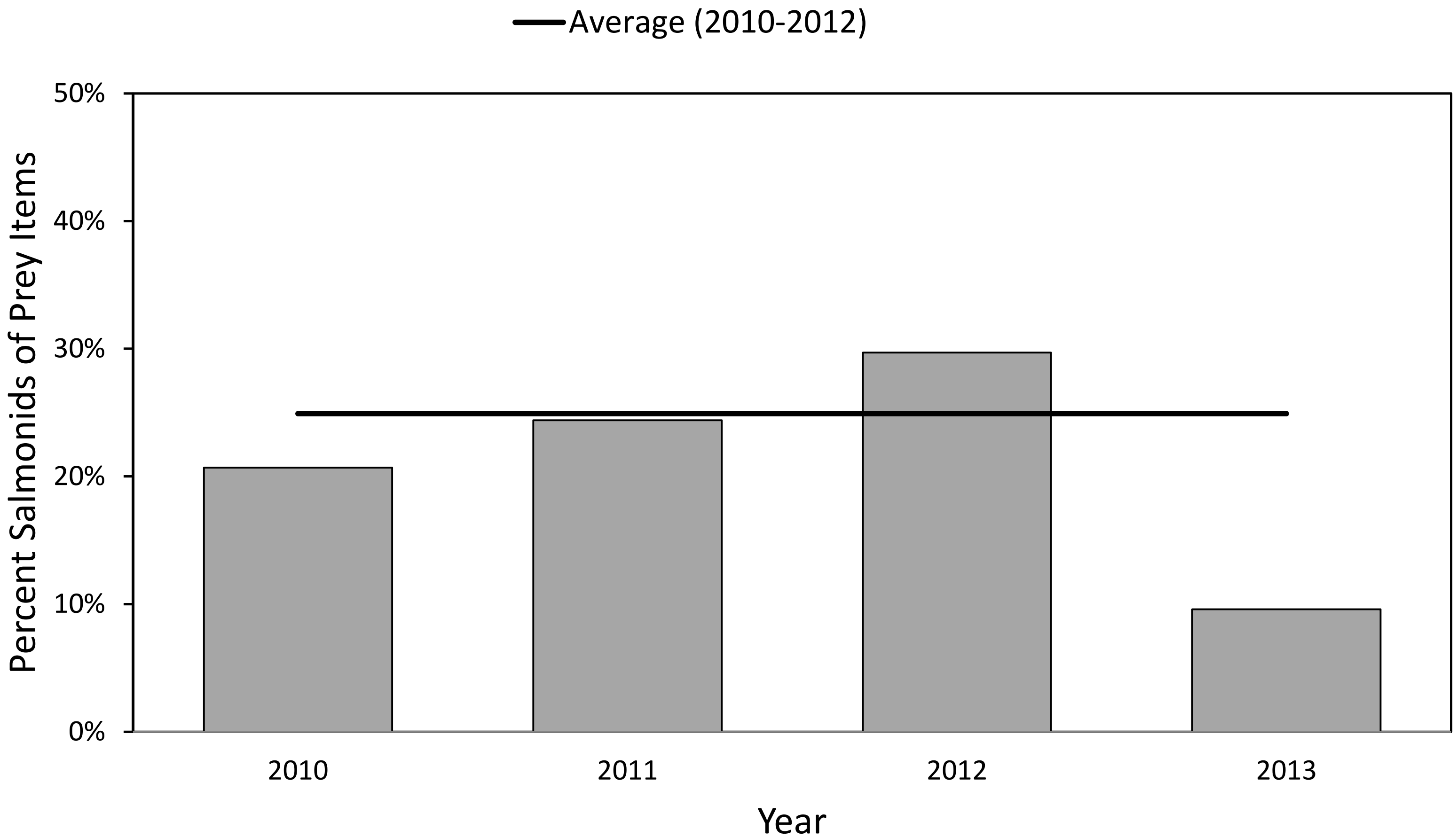


Figure 43. Average annual proportion of juvenile salmonids in the diet (percent of prey items) of Caspian terns nesting on Goose Island in Potholes Reservoir, Washington, during the 2010-2013 breeding seasons. Diet composition was based on fish identified in tern bill-loads on-colony.

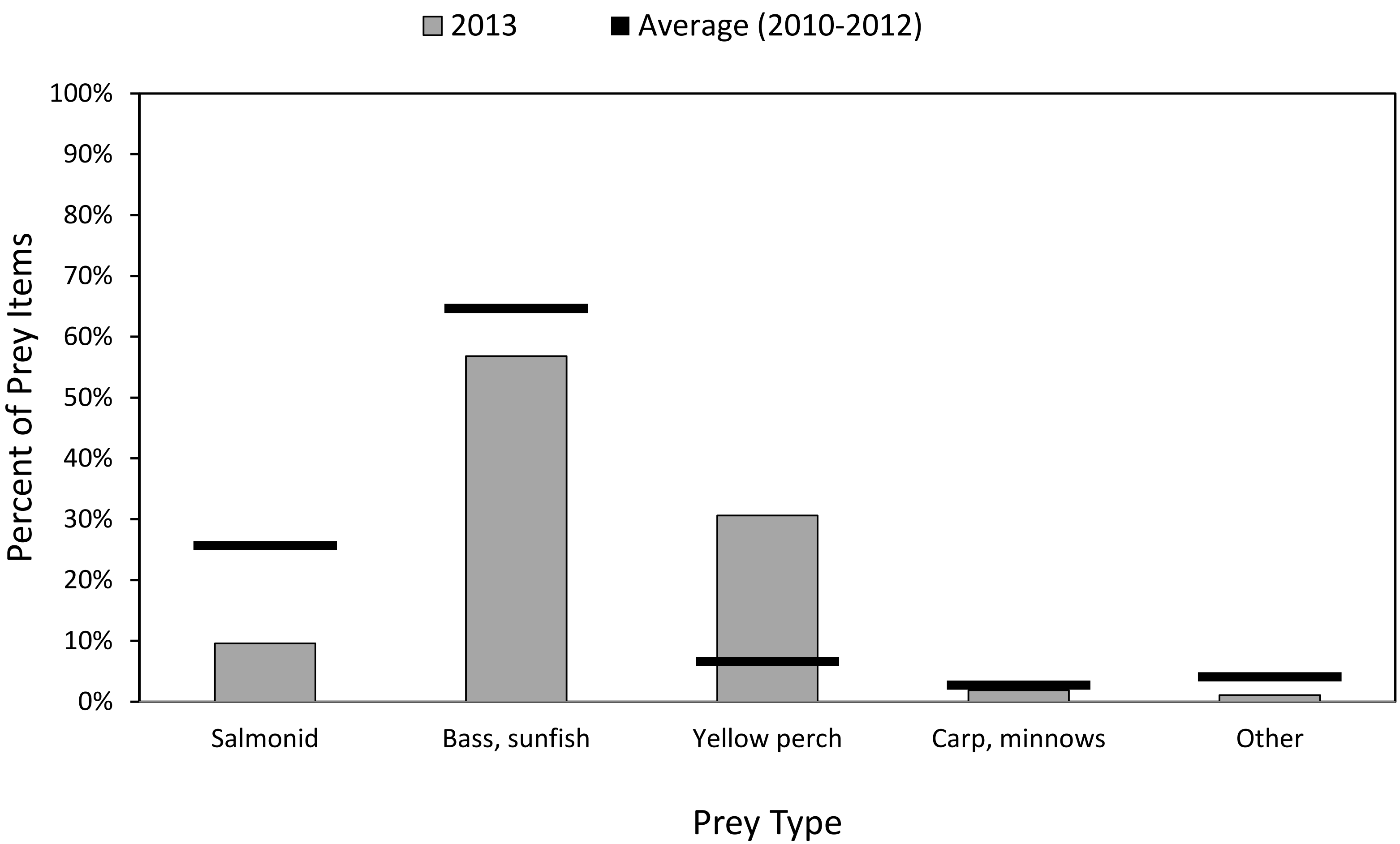


Figure 44. Diet composition (percent of prey items) of Caspian terns nesting on Goose Island in Potholes Reservoir, Washington, during the 2013 breeding season. Diet composition was based on fish identified in tern bill-loads on-colony.

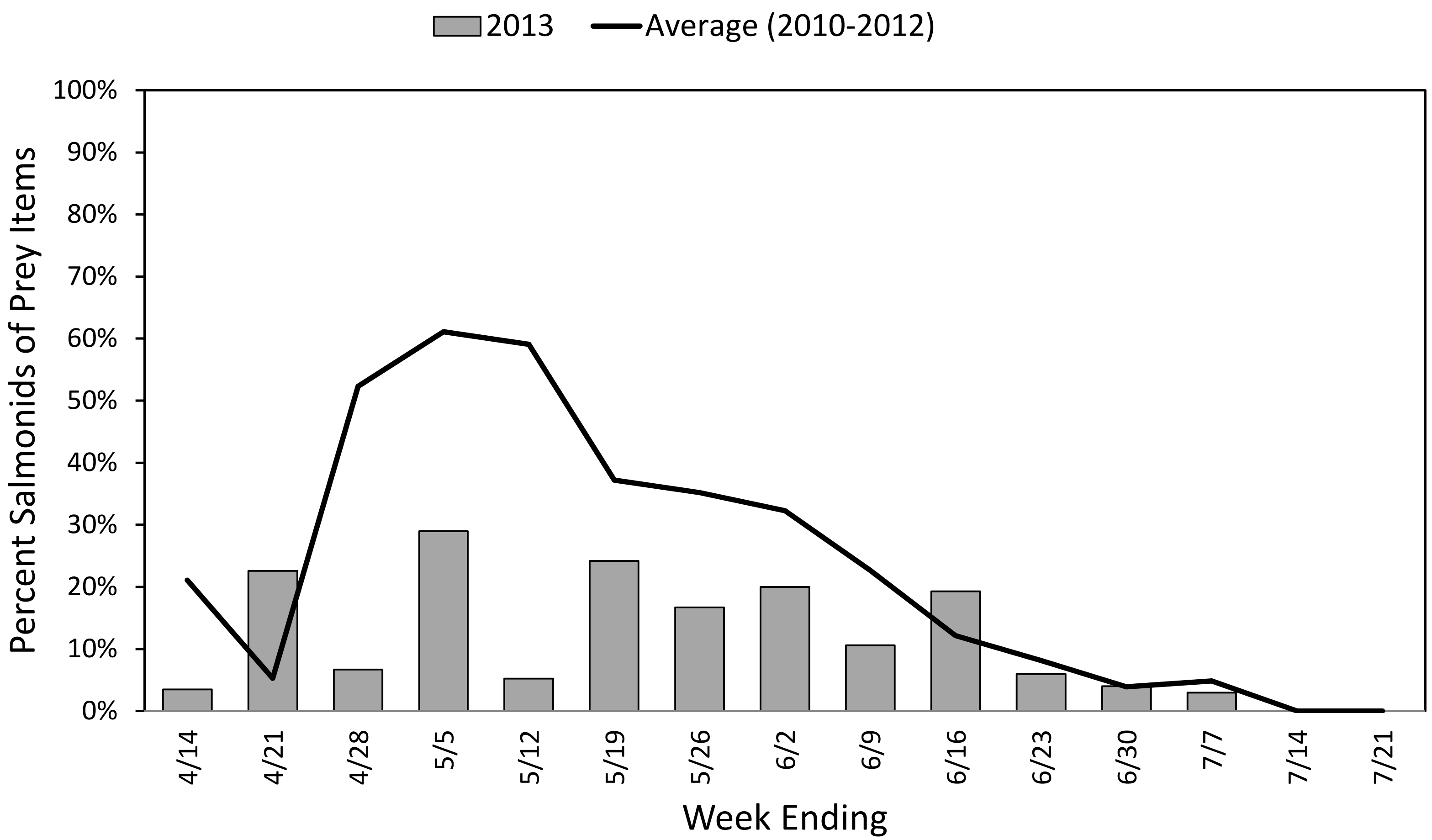


Figure 45. Proportion of juvenile salmonids in the diet (percent of prey items) of Caspian terns nesting on Goose Island in Potholes Reservoir, Washington, during the 2013 breeding season, by week. Diet composition was based on fish identified in tern bill-loads on-colony.

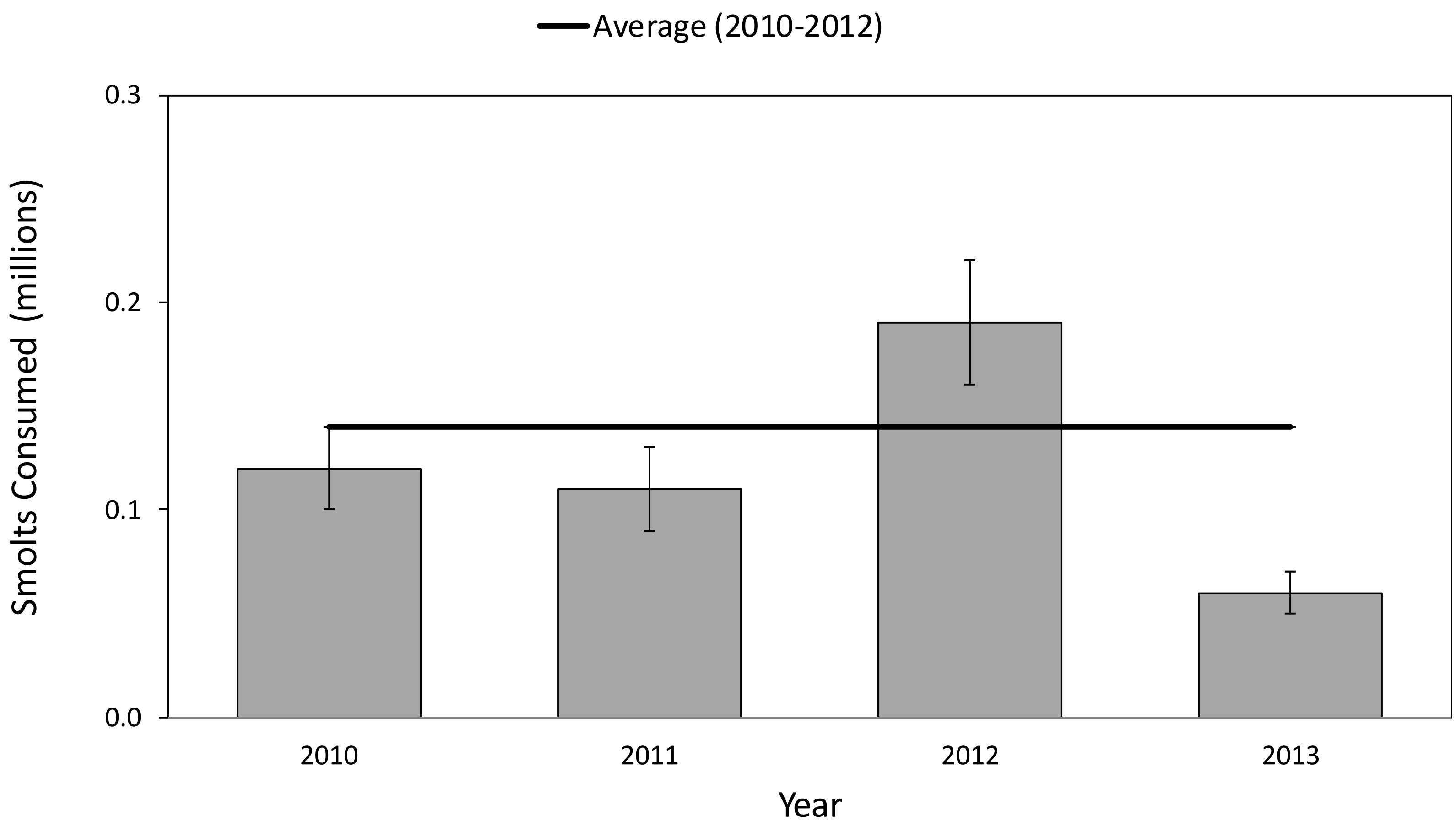


Figure 46. Estimated minimum total annual consumption of juvenile salmonids by Caspian terns nesting on Goose Island in Potholes Reservoir, Washington, during the 2010-2013 breeding seasons. Estimates are based on fish identified in tern bill-loads on-colony and bioenergetics calculations. Error bars represent 95% confidence intervals for the number of smolts consumed.

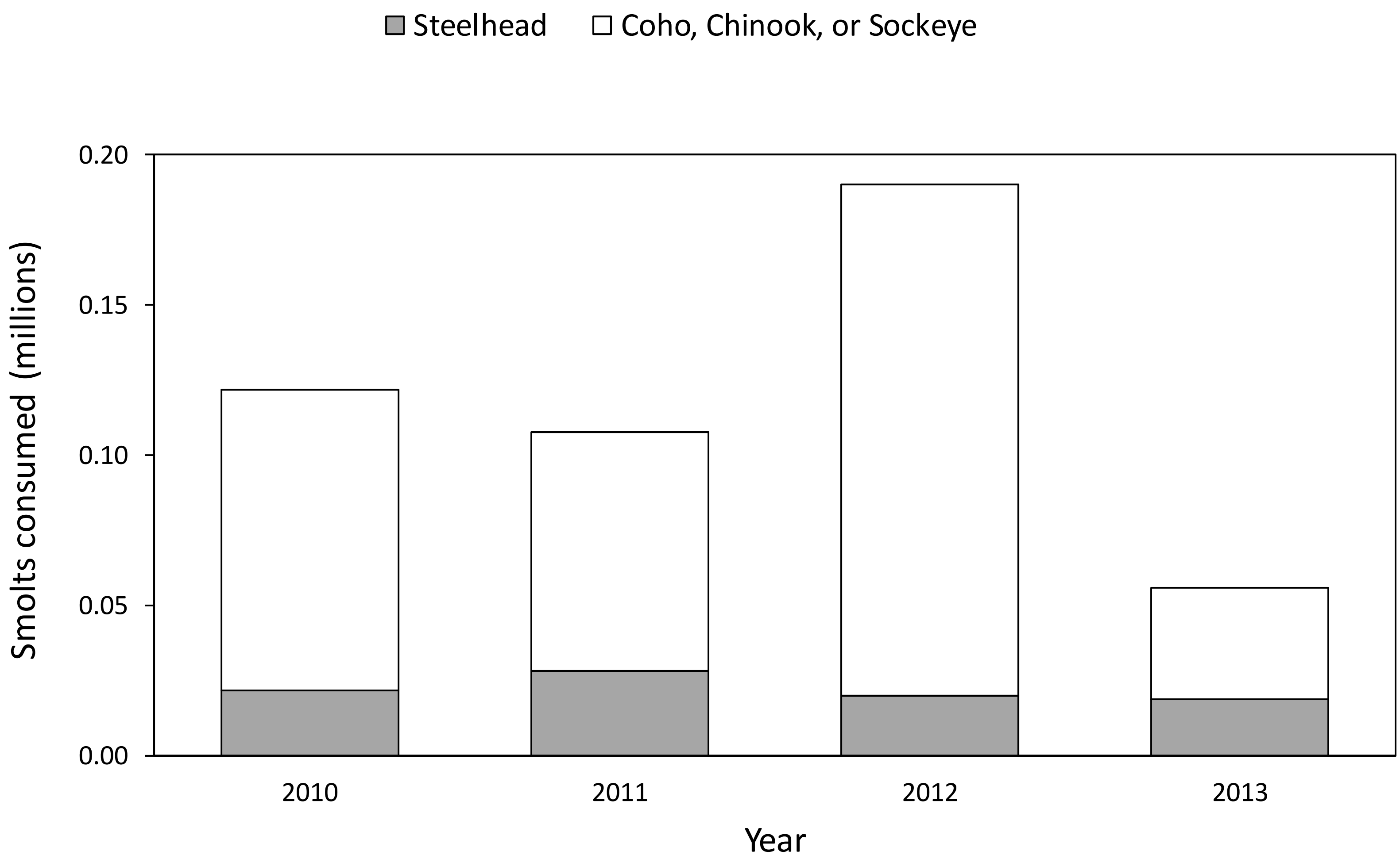


Figure 47. Estimated minimum total annual consumption of steelhead and salmon (coho, Chinook, or sockeye) smolts by Caspian terns nesting on Goose Island in Potholes Reservoir, Washington, during the 2010-2013 breeding seasons. Estimates are based on fish identified in tern bill-loads on-colony and bioenergetics calculations.

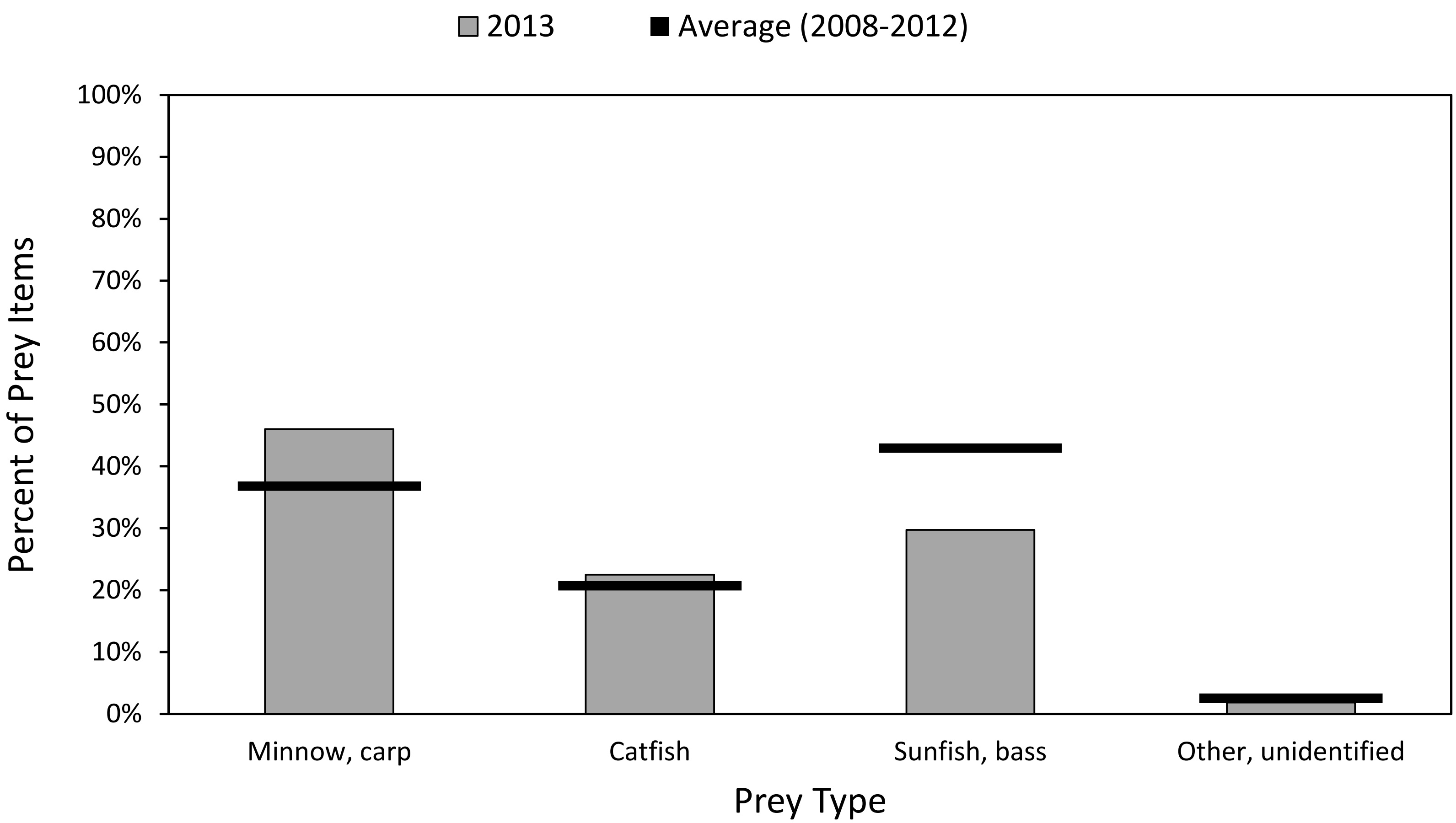


Figure 48. Diet composition (percent of prey items) of Caspian terns nesting on the Corps-constructed tern island at Crump Lake in Warner Valley, Oregon, during the 2013 breeding season. Diet composition was based on fish identified in tern bill-loads on-colony.

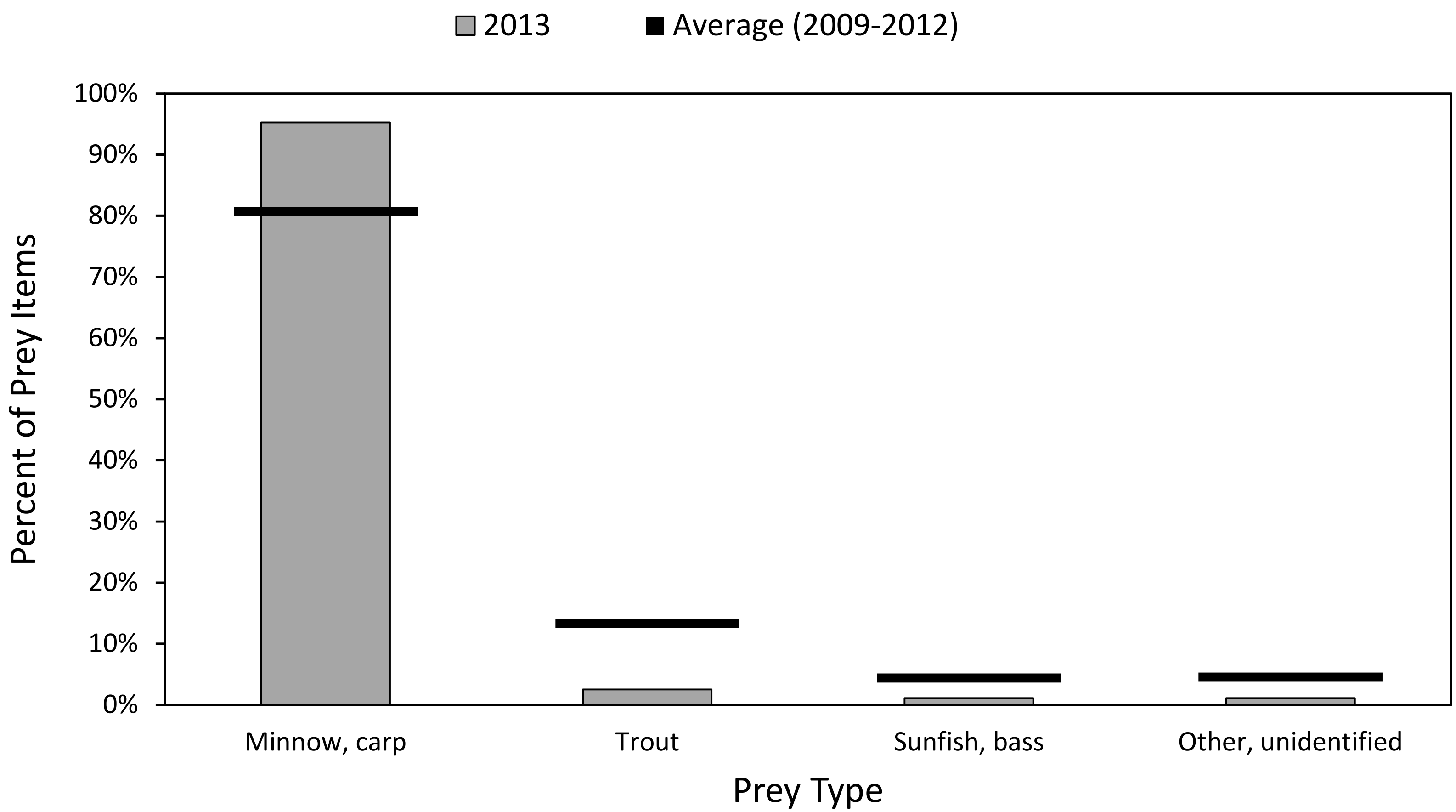


Figure 49. Diet composition (percent of prey items) of Caspian terns nesting on the Corps-constructed tern islands at Summer Lake Wildlife Area (East Link and Gold Dike) during the 2013 breeding season. Diet composition was based on fish identified in tern bill-loads on-colony.

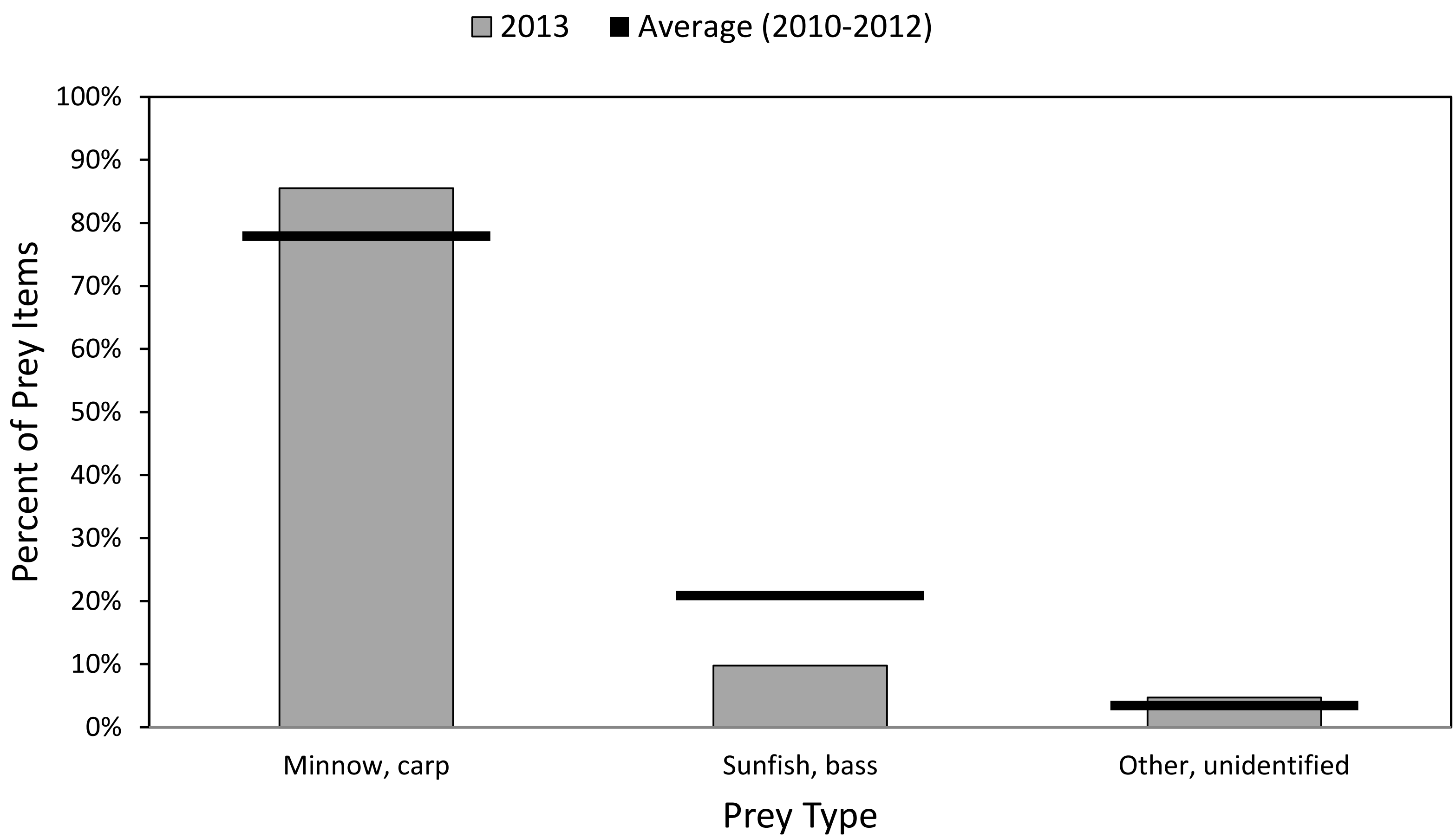


Figure 50. Diet composition (percent of prey items) of Caspian terns nesting on the Corps-constructed tern island at Sheepy Lake in Lower Klamath NWR, California, during the 2013 breeding season. Diet composition was based on fish identified in tern bill-loads on-colony.

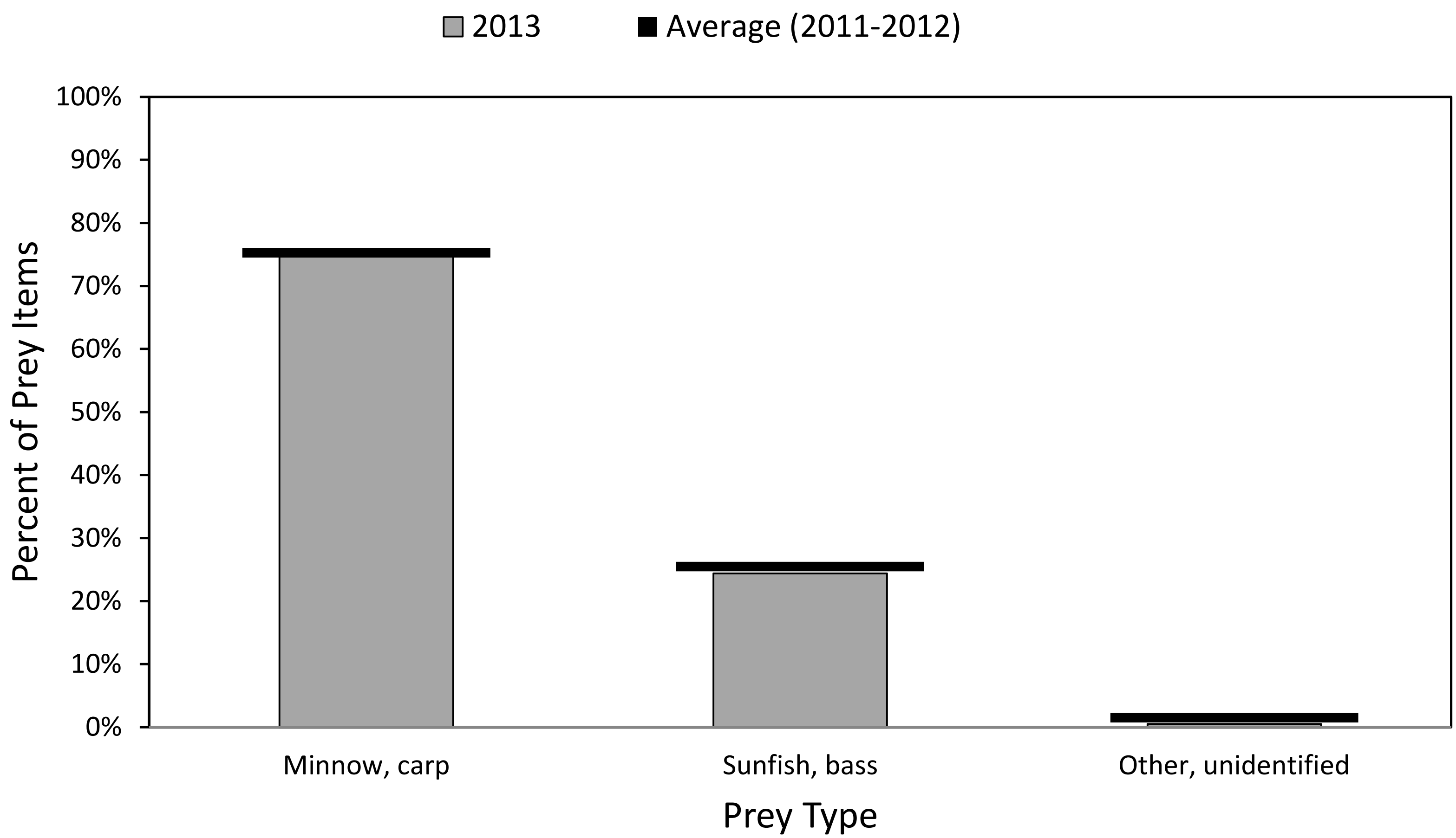


Figure 51. Diet composition (percent of prey items) of Caspian terns nesting on the Corps-constructed tern island at Tule Lake Sump 1B in Tule Lake NWR, California during the 2013 breeding season. Diet composition was based on fish identified in tern bill-loads on-colony.

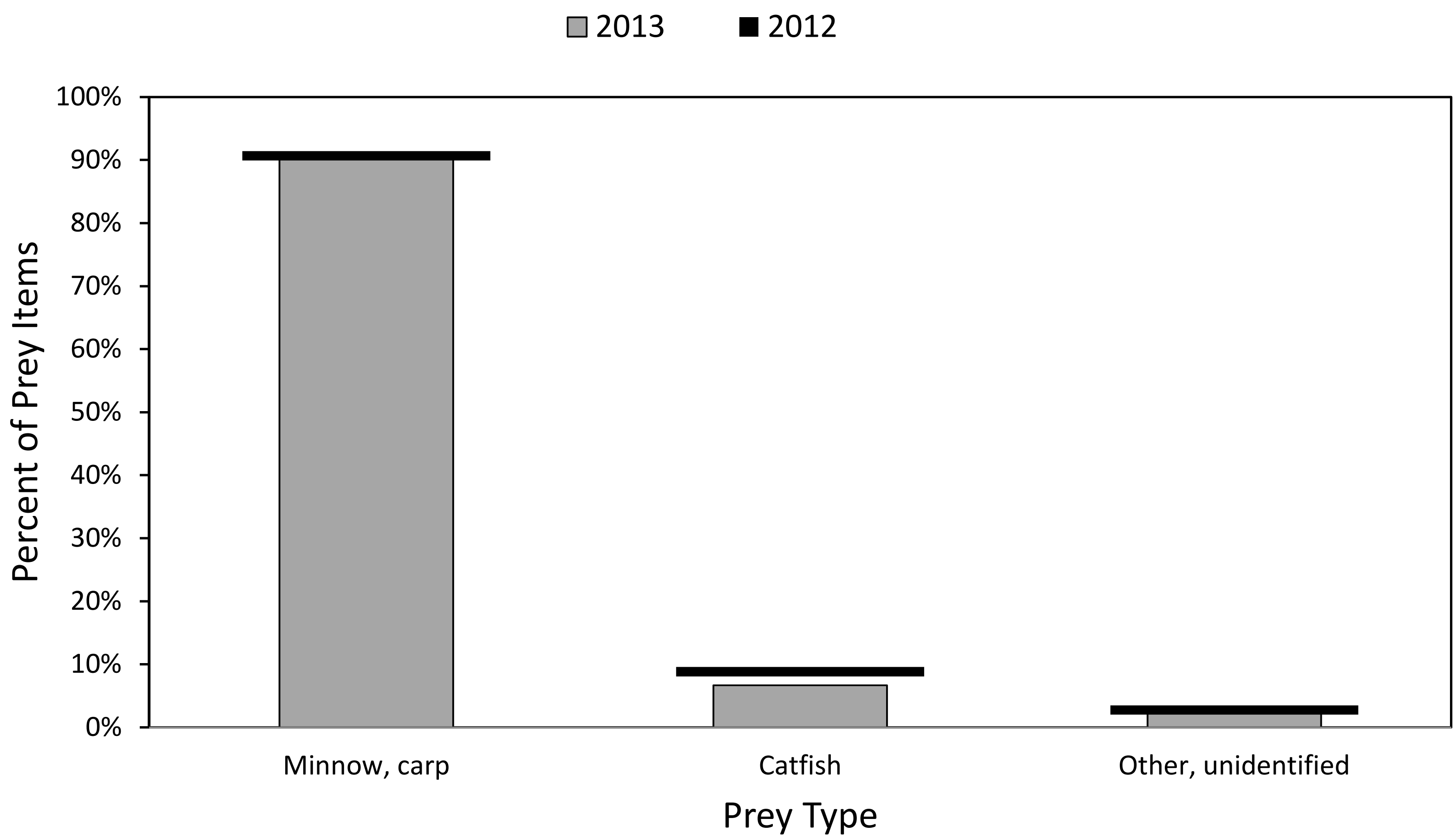


Figure 52. Diet composition (percent of prey items) of Caspian terns nesting on the Corps-constructed tern island at Malheur Lake in Malheur NWR, Oregon, during the 2013 breeding season. Diet composition was based on fish identified in tern bill-loads on-colony.

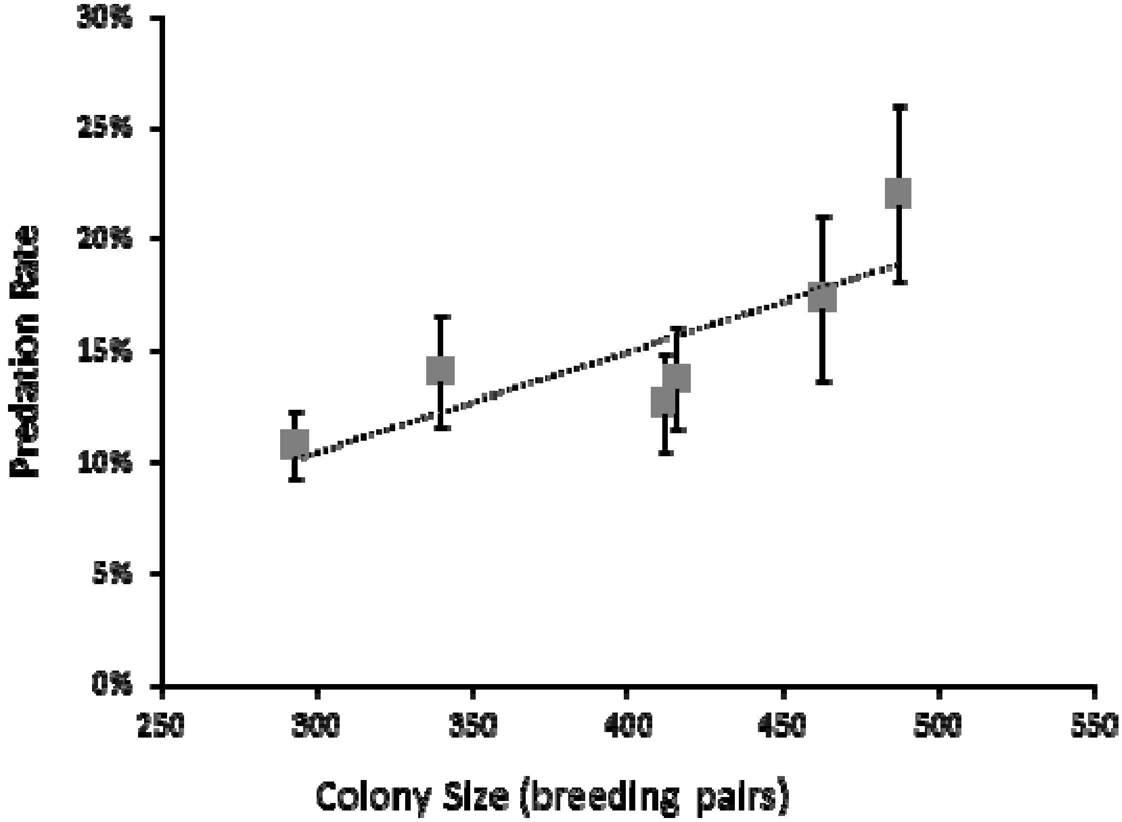


Figure 53. Relationship between annual Goose Island Caspian tern colony size and predation rates on upper Columbia River steelhead released from Rock Island Dam during 2008-2013. Dotted line indicates the best linear fit to the data.

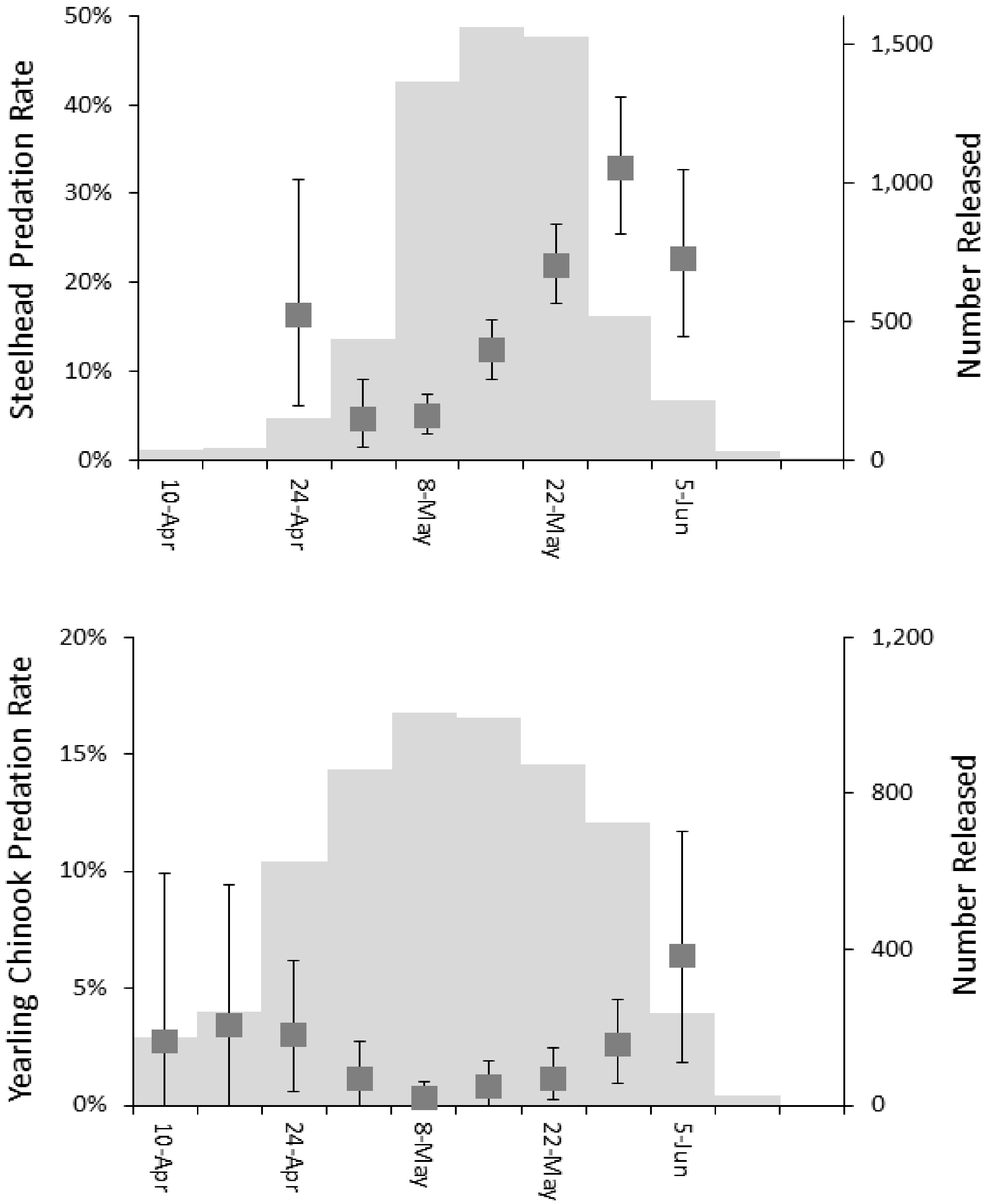


Figure 54. Weekly predation rates on PIT-tagged Upper Columbia River steelhead and yearling Chinook salmon released at Rock Island Dam by Caspian terns nesting on Goose Island, Potholes Reservoir during 2013. Only weeks when more than 100 PIT-tagged steelhead were released from Rock Island Dam are shown. Error bars represent 95% confidence intervals. The numbers of smolts PIT-tagged and released at Rock Island Dam per week are also shown.

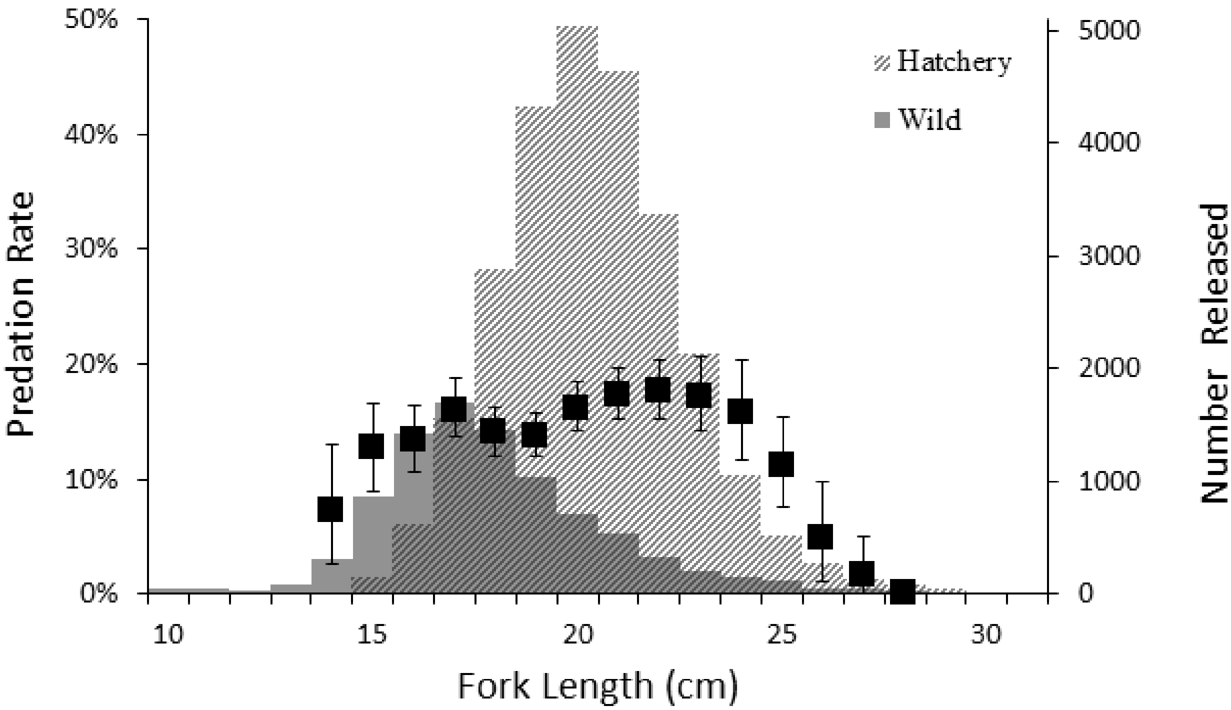


Figure 55. Fork length-specific predation rates of PIT-tagged Upper Columbia River steelhead released at Rock Island Dam by Caspian terns nesting at Goose Island, Potholes Reservoir during 2008-2012 (boxes). Only fork length groups with more than 100 PIT-tagged steelhead released from Rock Island Dam per year are shown. Error bars represent 95% confidence intervals. The numbers of hatchery (thatched) and wild (grey) steelhead smolts PIT-tagged and released at Rock Island Dam per fork length category are also shown.

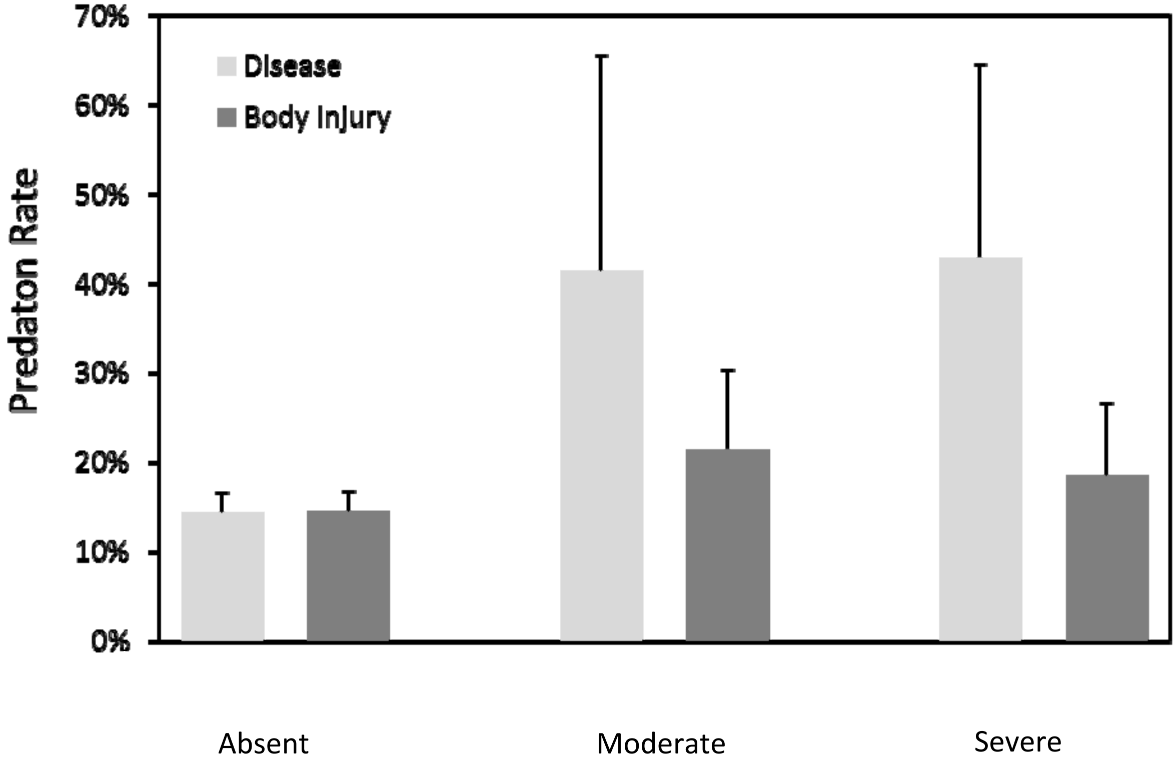


Figure 56. Predation rates of PIT-tagged Upper Columbia River steelhead (released at Rock Island Dam) by Caspian terns nesting at Goose Island, Potholes Reservoir during 2013 separated by levels of body injury and disease (absent, moderate, severe). Absent refers to the lack of external symptoms (n = 5,784 without disease; n = 5,431 without body injuries), moderate refers to infection limited to one area or small and/or healed injuries (n = 51 with moderate disease; n = 235 with moderate injuries), and severe refers to infections in multiple areas or open wounds and/or injuries covering a large surface area of the body (n = 58 with severe disease; n = 227 with severe injuries). Error bars represent 95% confidence intervals.

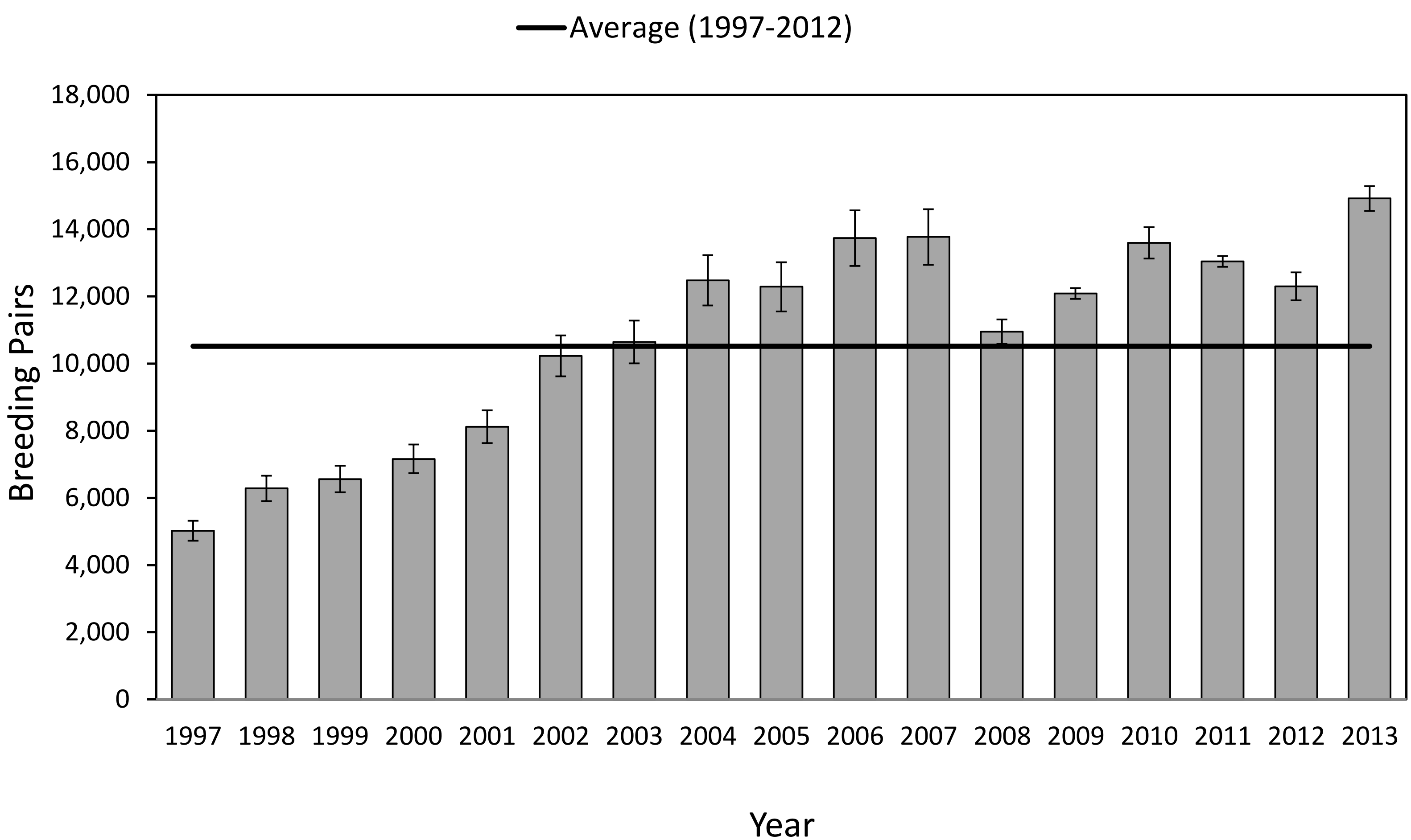


Figure 57. Size of the double-crested cormorant breeding colony (number of breeding pairs) on East Sand Island in the Columbia River estuary during the 1997-2013 breeding seasons. Error bars represent 95% confidence intervals for the number of breeding pairs.

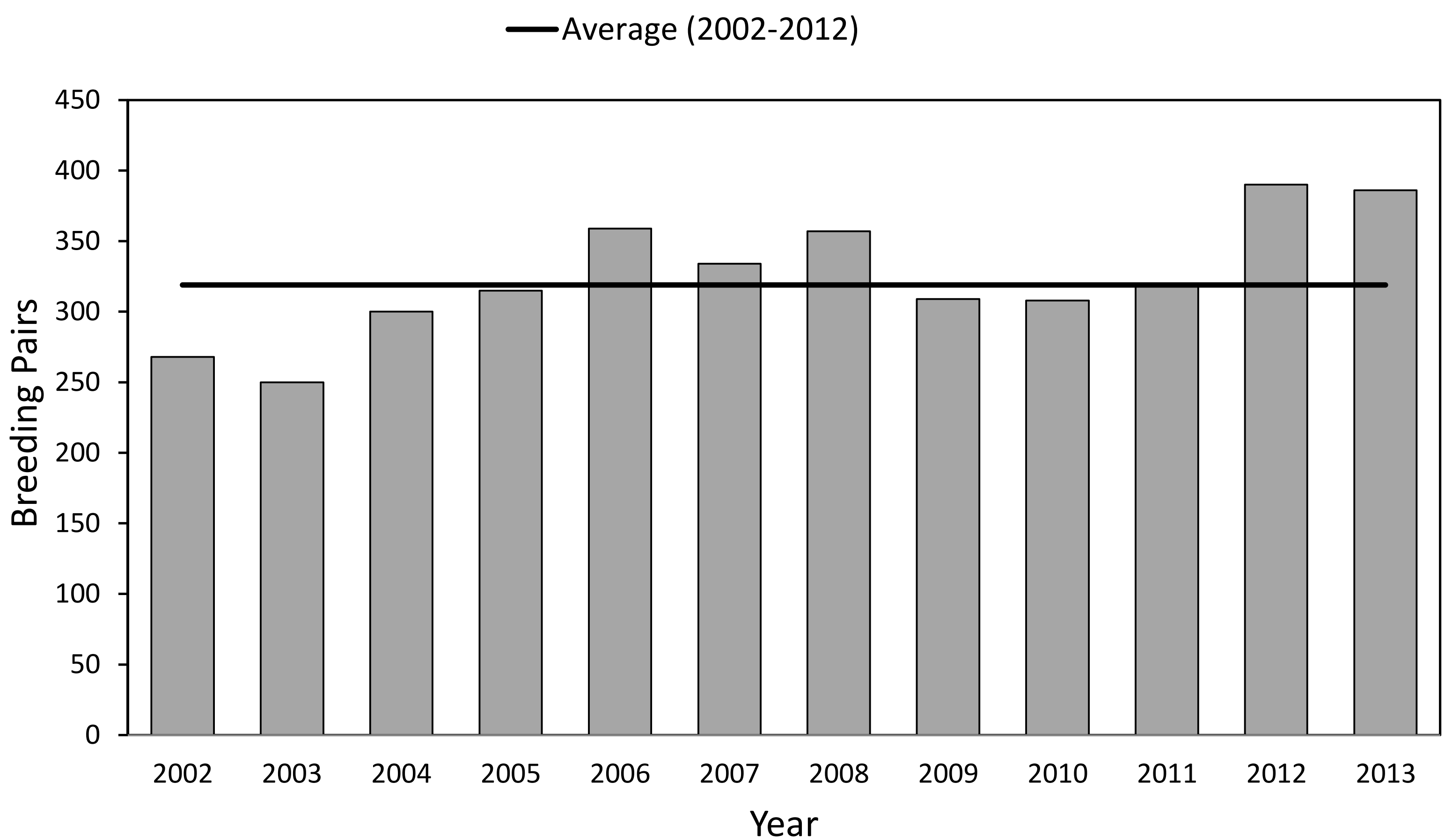


Figure 58. Size of the double-crested cormorant breeding colony (number of breeding pairs) on Foundation Island in the mid-Columbia River during the 2002-2013 breeding seasons.

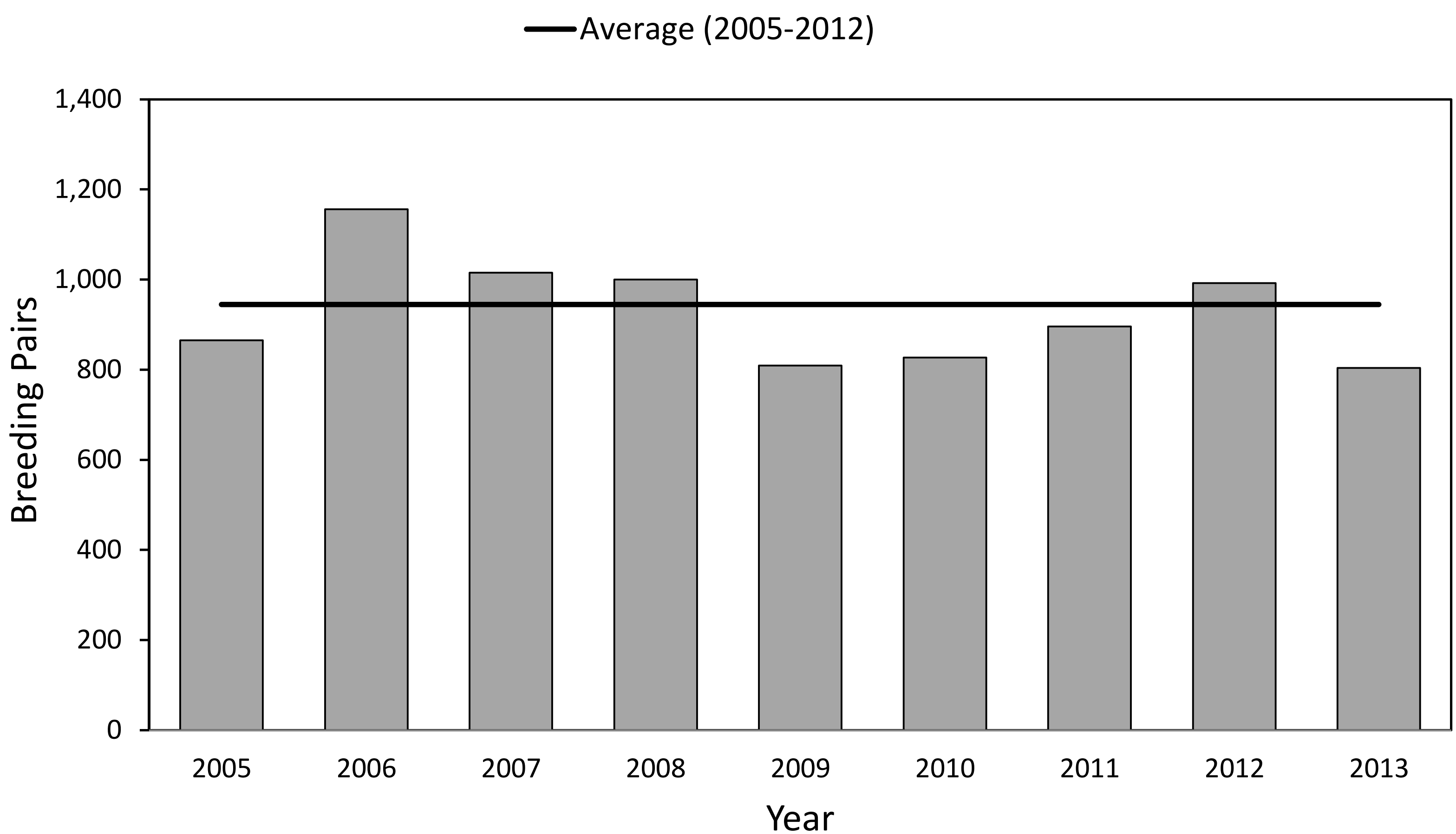


Figure 59. Estimated size of the double-crested cormorant breeding colony (number of breeding pairs) in North Potholes Reserve, Potholes Reservoir, Washington, during the 2005-2013 breeding seasons.

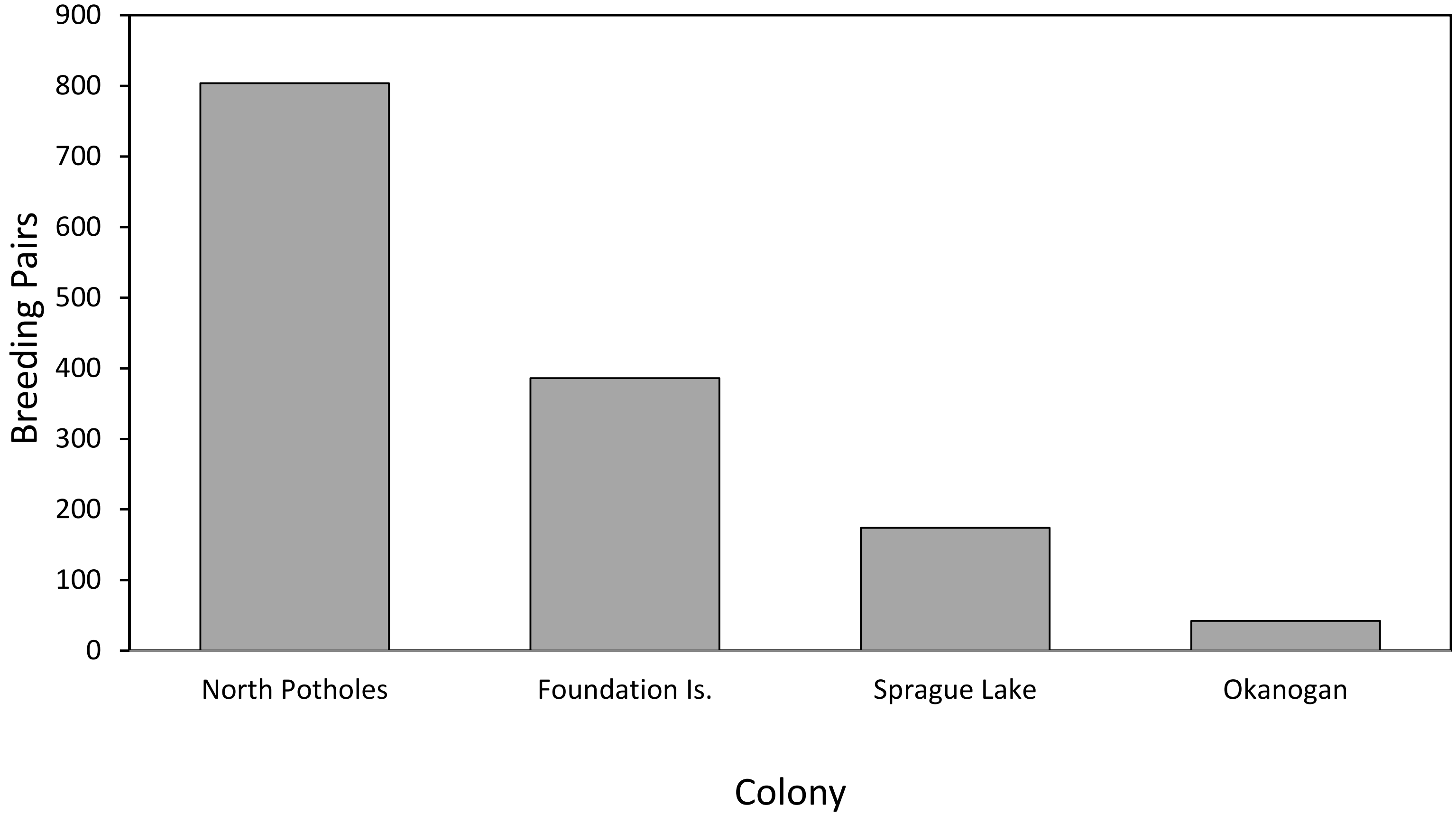


Figure 60. Size of the double-crested cormorant breeding colonies (number of breeding pairs) in the Columbia Plateau region during the 2013 breeding season.

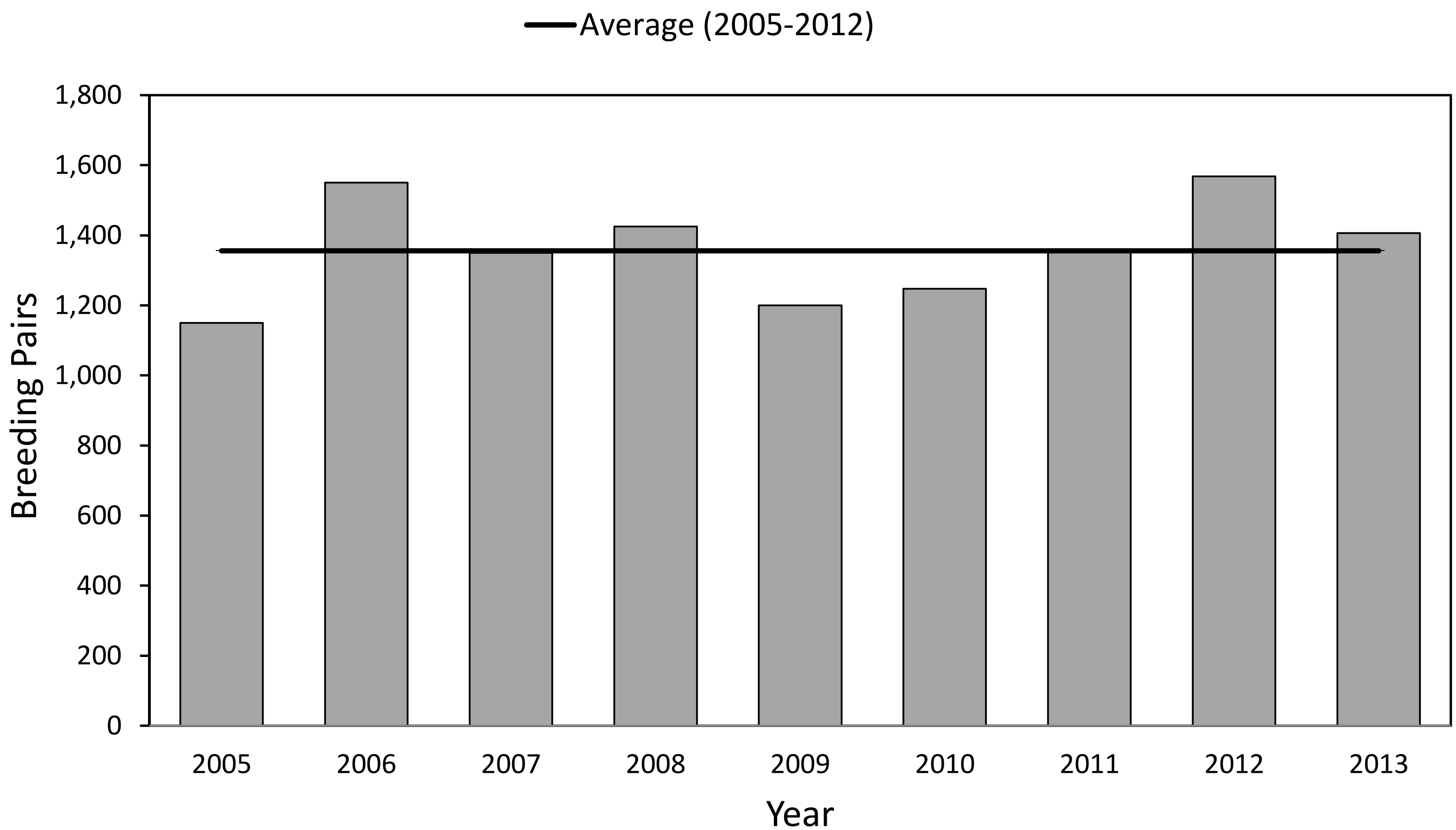


Figure 61. Estimated total number of breeding pairs of double-crested cormorant nesting at colonies in the Columbia Plateau region during the 2005-2013 breeding seasons.

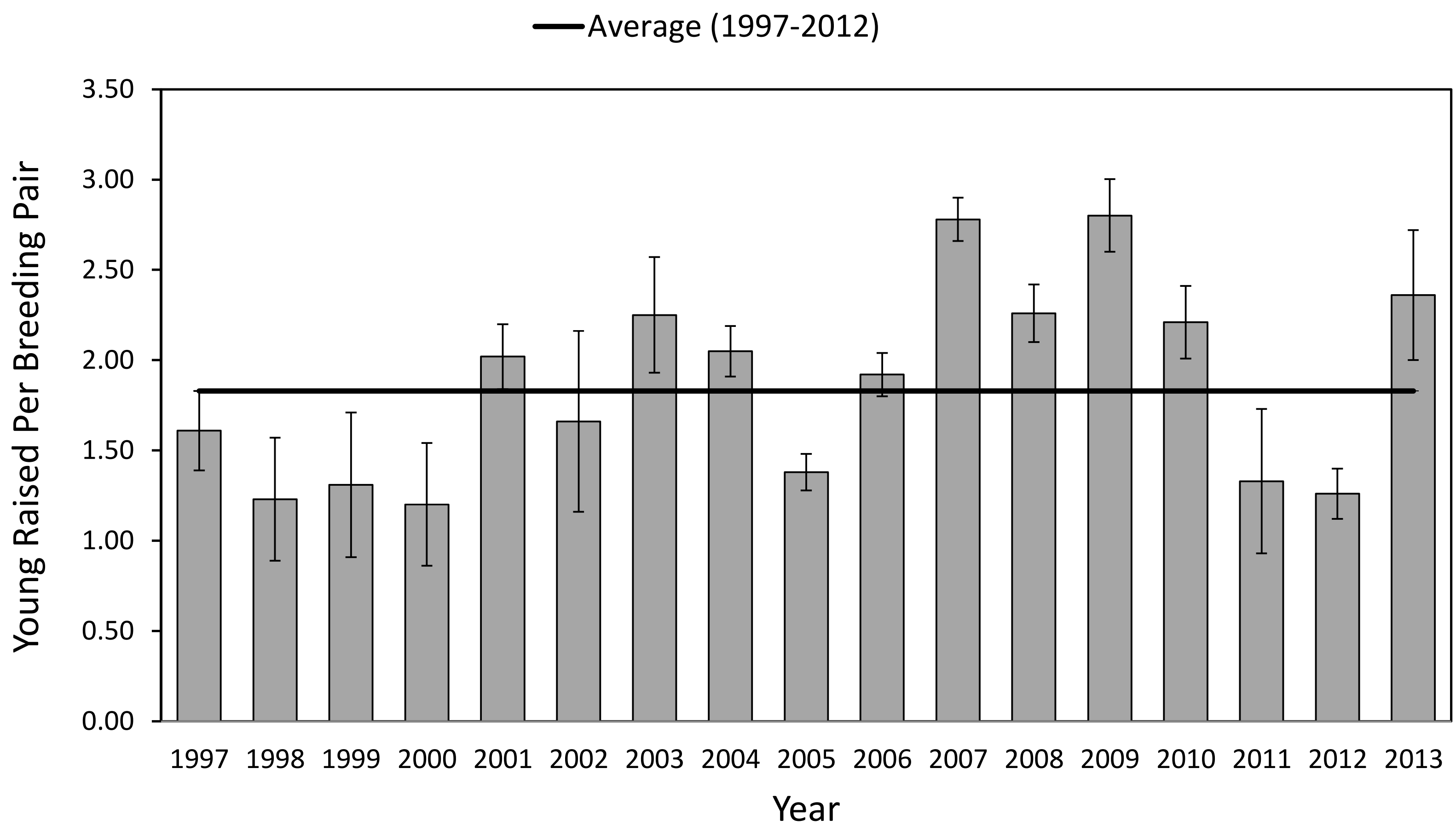


Figure 62. Double-crested cormorant nesting success (average number of young raised per breeding pair) at the colony on East Sand Island in the Columbia River estuary during the 1997-2013 breeding seasons. Error bars represent 95% confidence intervals.

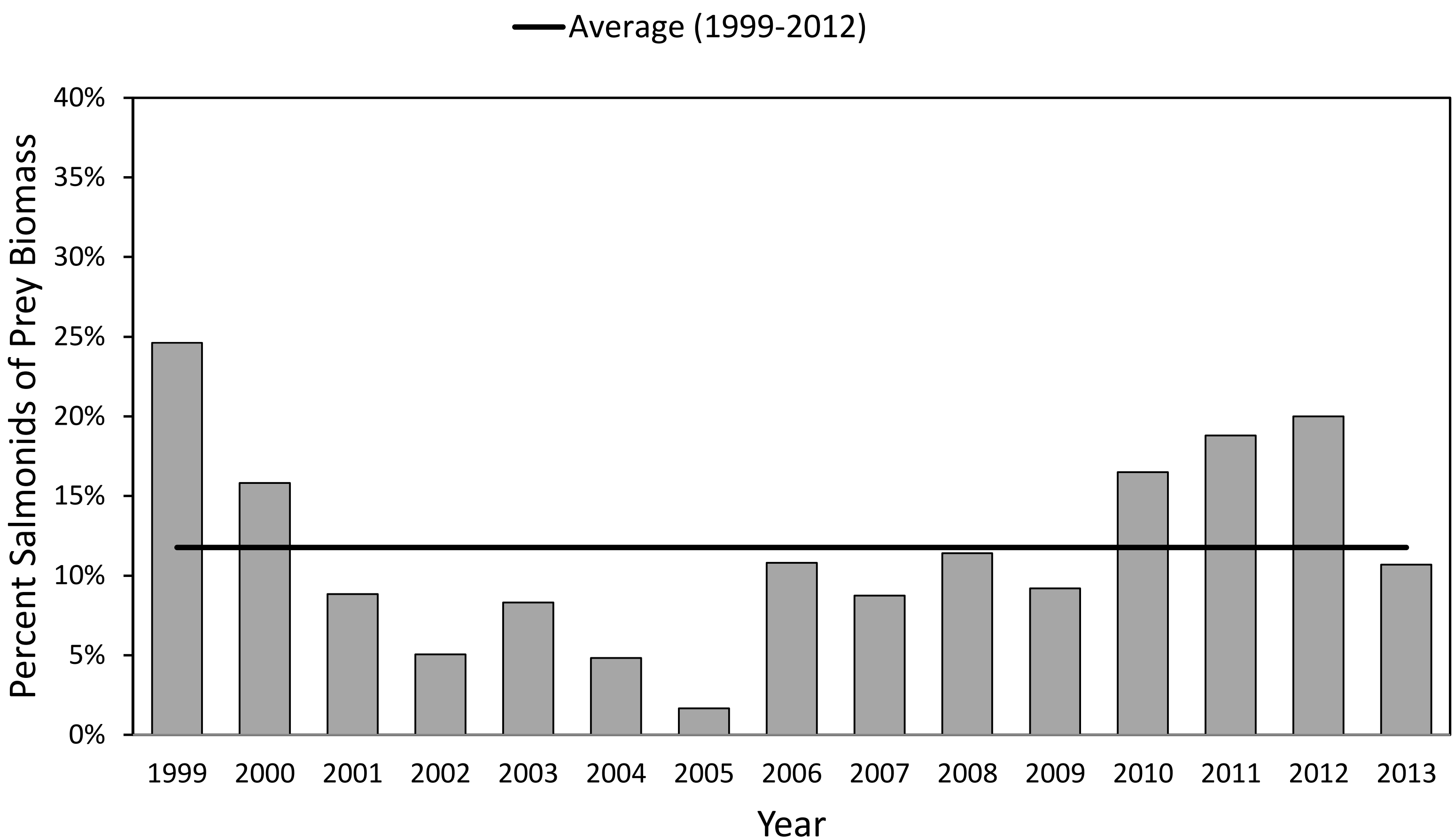


Figure 63. Average annual proportion of juvenile salmonids in the diet (percent of prey biomass) of double-crested cormorants nesting on East Sand Island in the Columbia River estuary during the 1999-2013 breeding seasons. Diet composition is based on analysis of stomach contents samples collected near the cormorant colony.

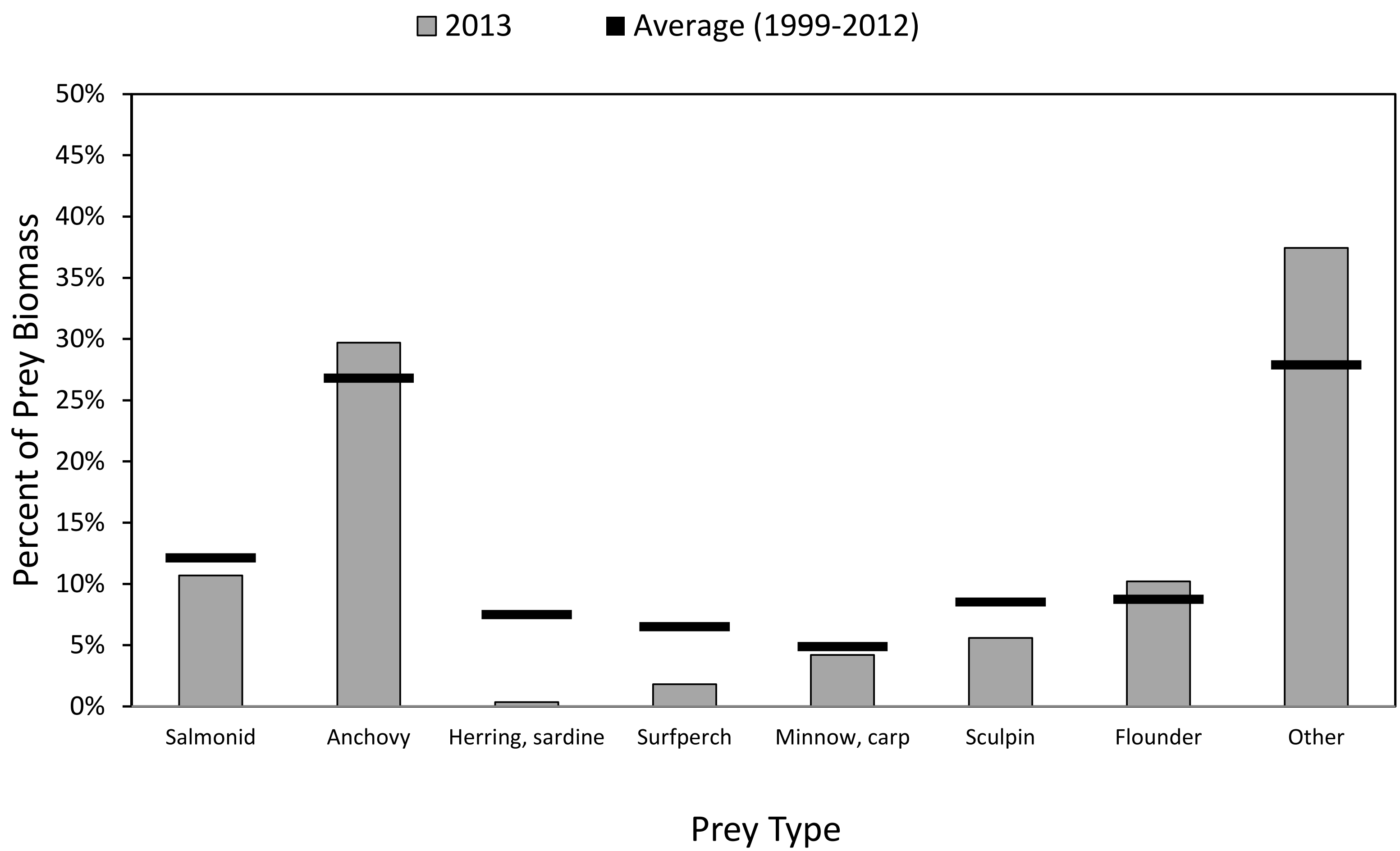


Figure 64. Diet composition (percent of prey biomass) of double-crested cormorants nesting on East Sand Island in the Columbia River estuary during the 2013 breeding season. Diet composition was based on fish identified in cormorant foregut samples.

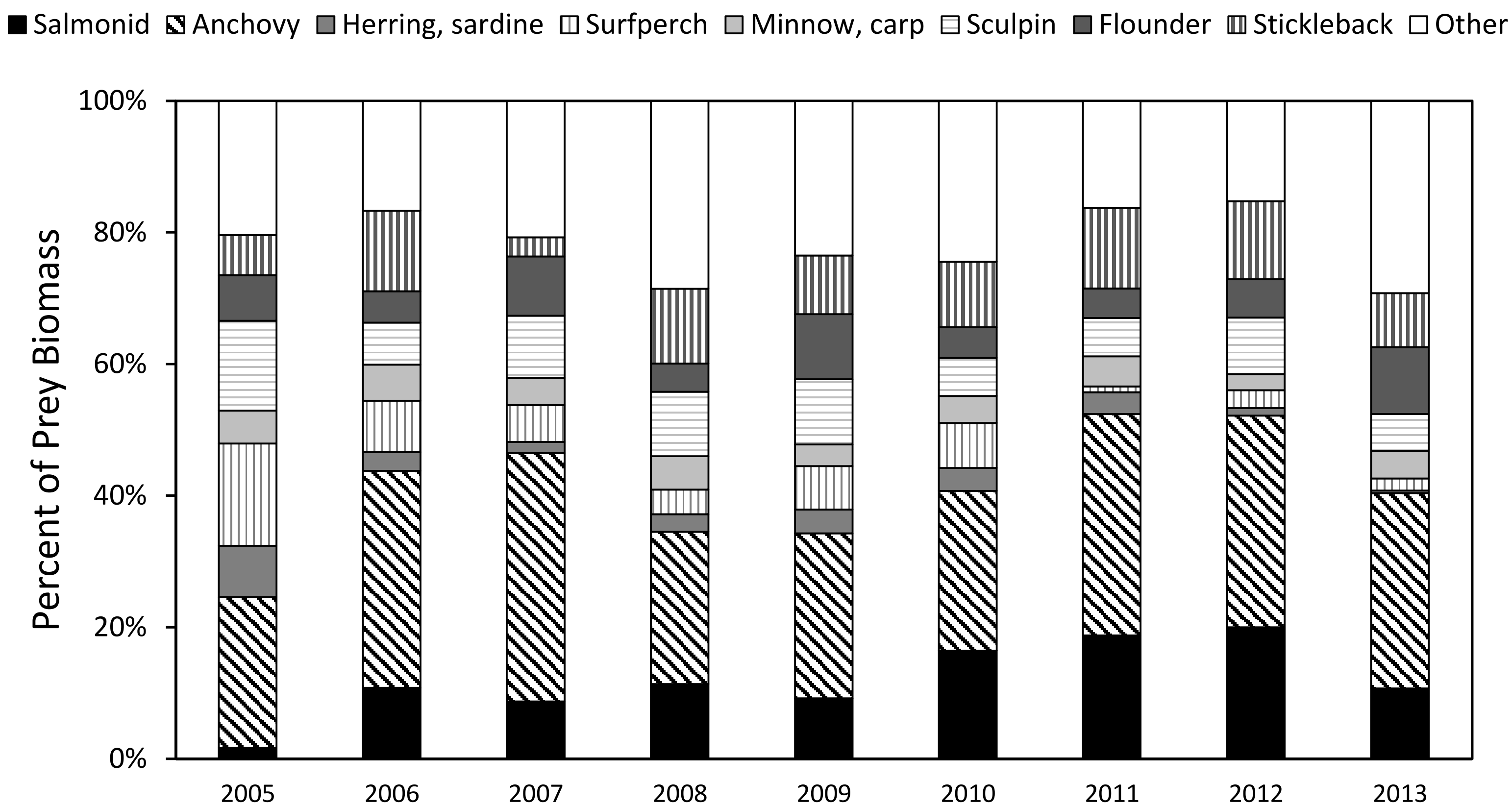


Figure 65. Annual diet composition (percent of prey biomass) of double-crested cormorants nesting on East Sand Island in the Columbia River estuary during the 2005-2013 breeding seasons. Diet composition was based on fish identified in cormorant foregut samples.

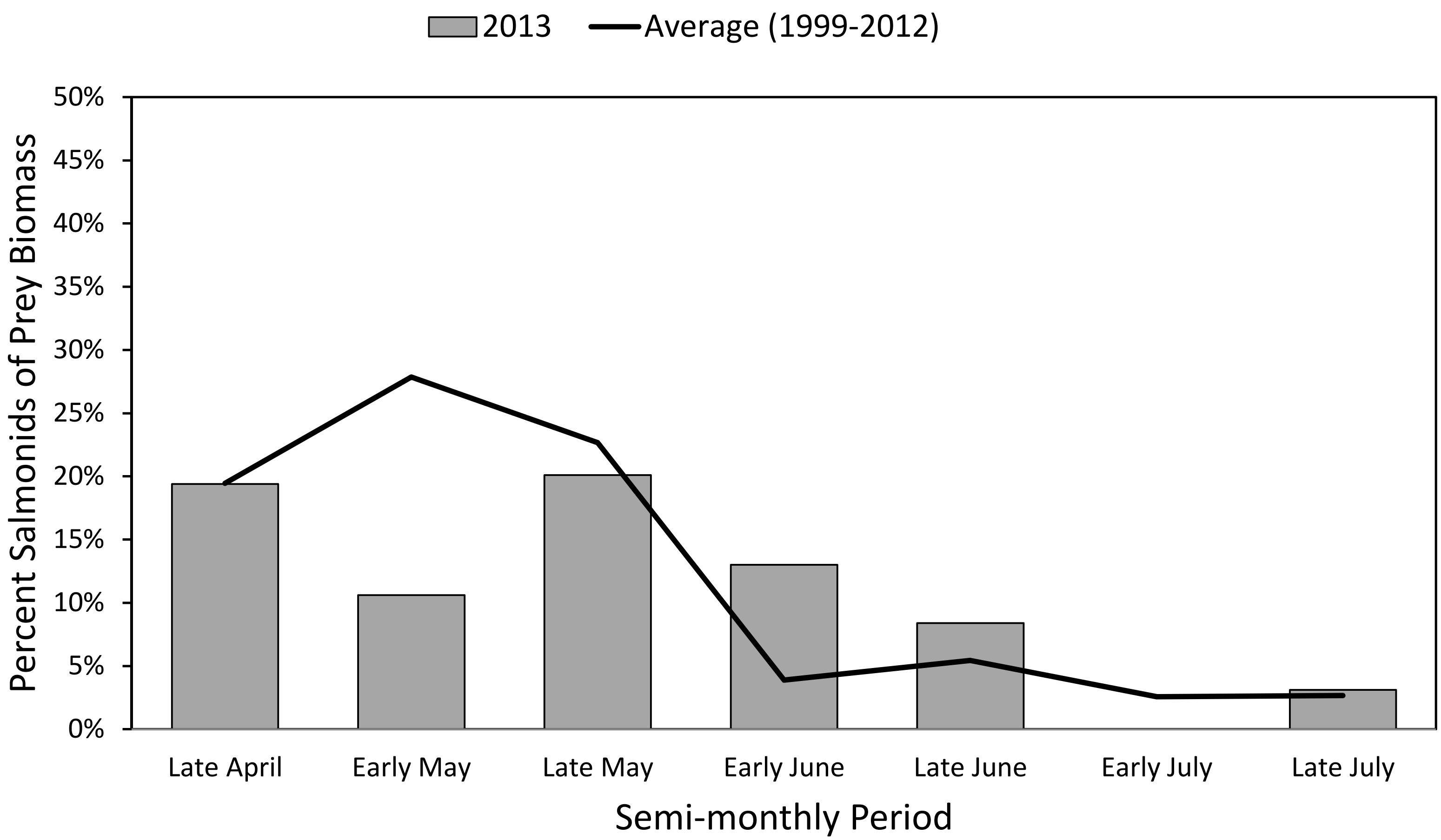
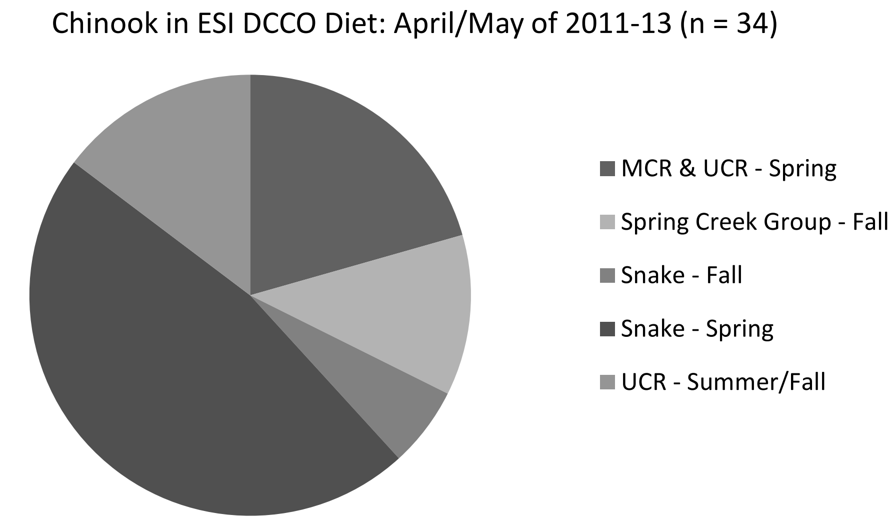


Figure 66. Seasonal trend in the proportion of juvenile salmonids in the diet (percent of prey biomass) of double-crested cormorants nesting on East Sand Island in the Columbia River estuary during the 2013 breeding season, by half-month period. Diet composition was based on fish identified in cormorant foregut samples.



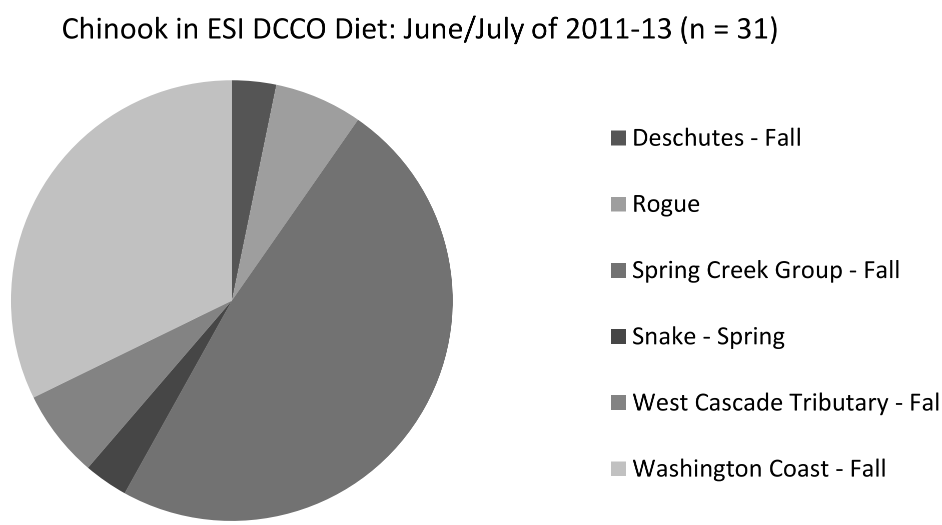
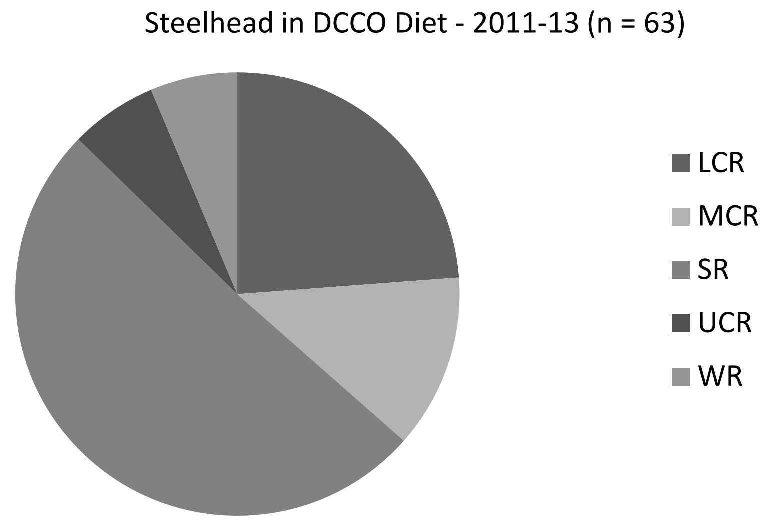


Figure 67. Genetic stock of origin for Chinook salmon in the diet of double-crested cormorants (DCCO) nesting on East Sand Island (ESI) in the Columbia River estuary. Genetic stock identification of salmonids was performed by D. Kuligowski, NOAA Fisheries, on salmonids in stomach contents samples collected from double-crested cormorants returning to the East Sand Island colony during the 2011-2013 breeding seasons. The Rogue River fall run stock was introduced to the lower Columbia River as part of a select area fishery enhancement project (North et al. 2006).



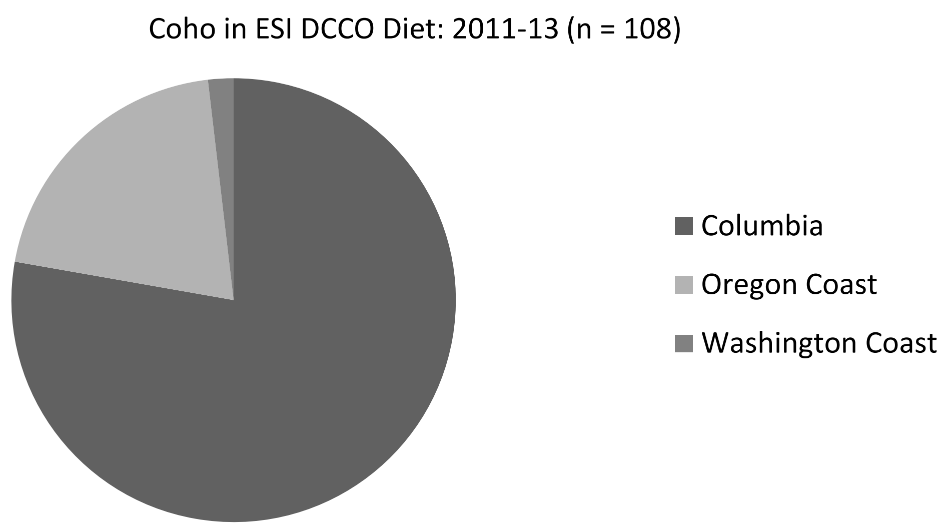


Figure 68. Genetic stock of origin for steelhead trout and coho salmon in the diet of double-crested cormorants (DCCO) nesting on East Sand Island (ESI) in the Columbia River estuary. Genetic stock identification of salmonids was performed by D. Kuligowski, NOAA Fisheries, on salmonids in stomach contents samples collected from double-crested cormorants returning to the East Sand Island colony during the 2011-2013 breeding seasons.

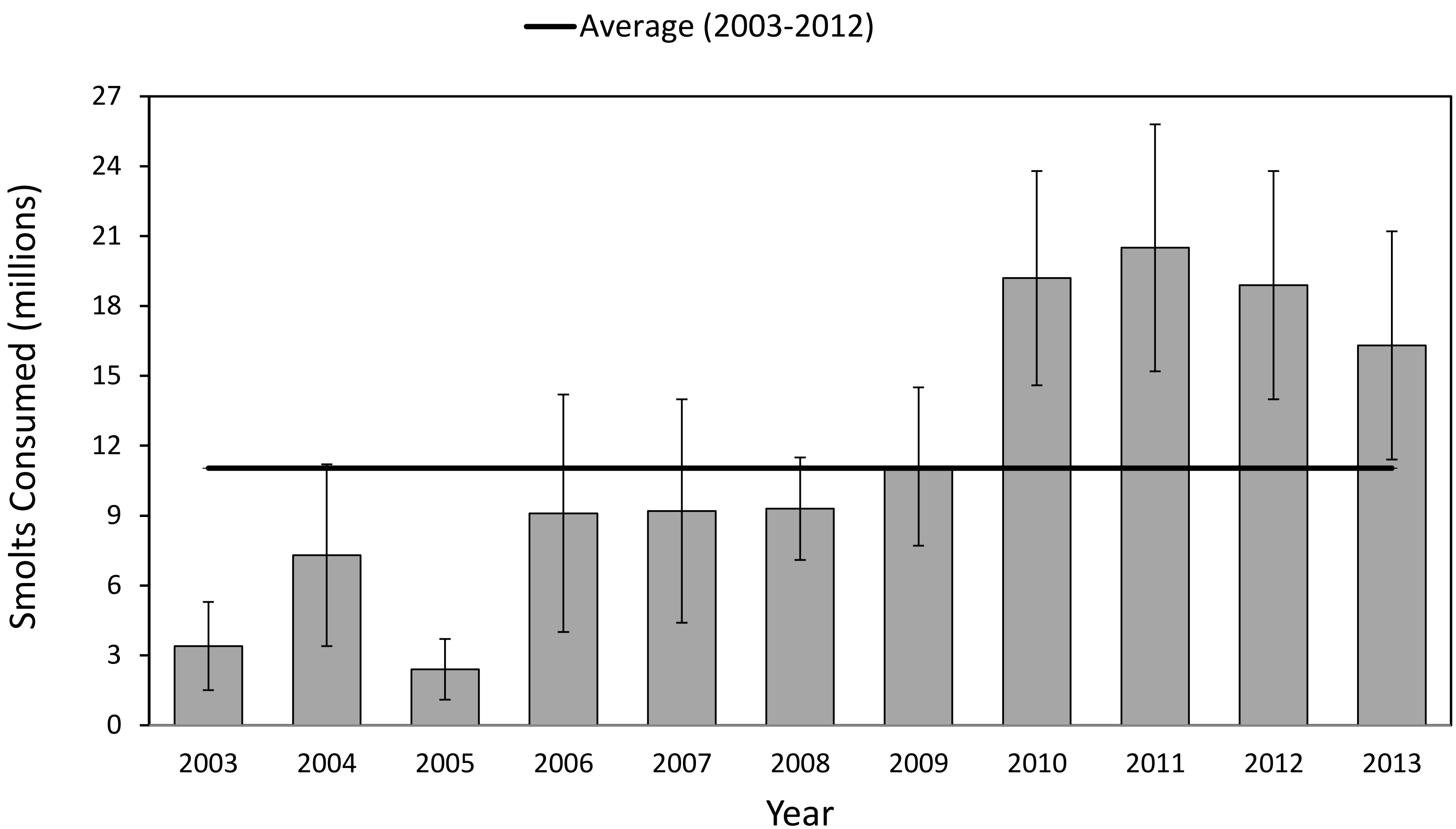


Figure 69. Estimated total annual consumption of juvenile salmonids by double-crested cormorants nesting on East Sand Island in the Columbia River estuary during the 2003-2013 breeding seasons. Estimates are based on fish identified in cormorant foregut samples collected near the colony and bioenergetics calculations. Error bars represent 95% confidence intervals for the number of smolts consumed.

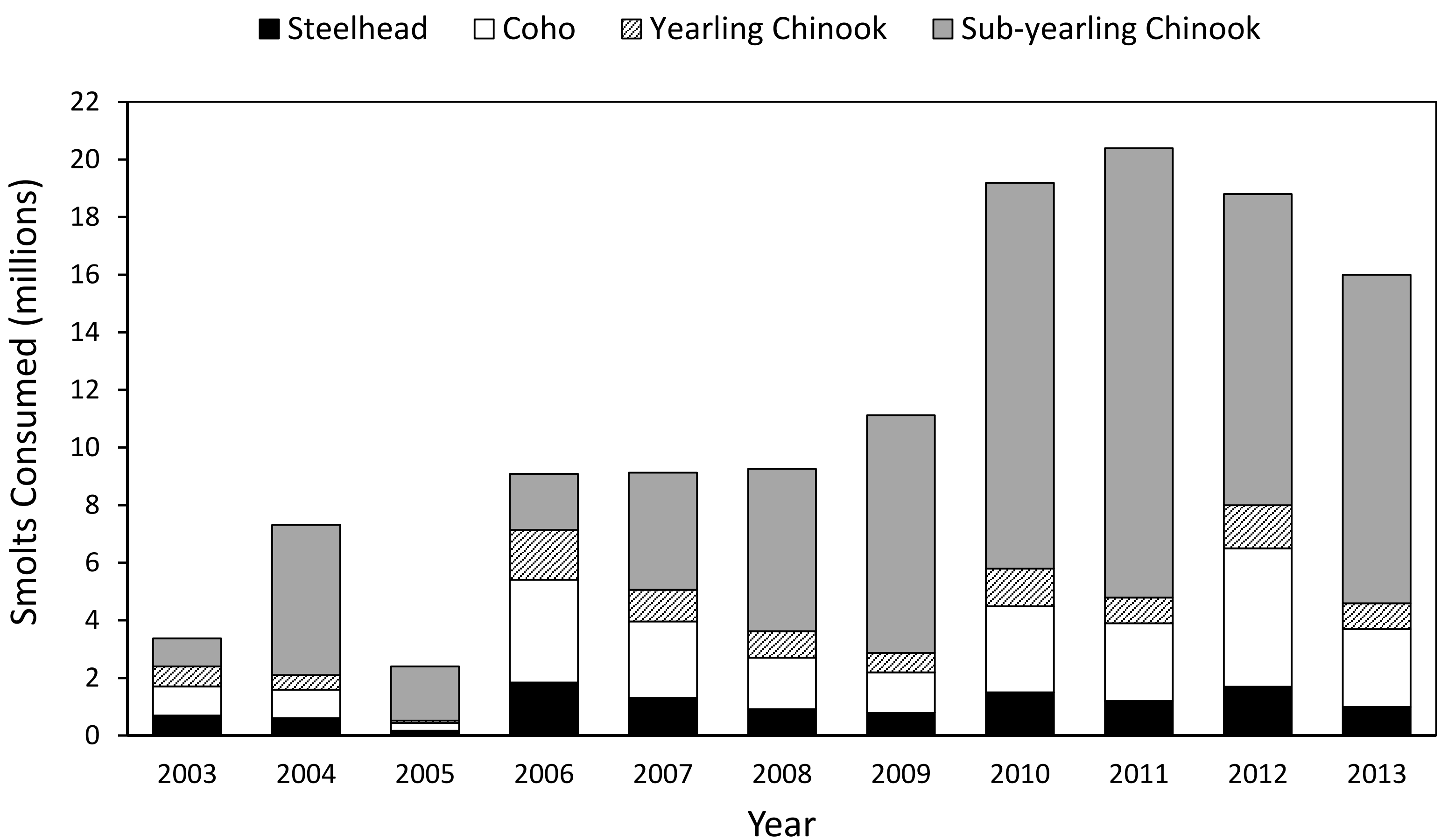


Figure 70. Estimated total annual consumption of four species/run types of juvenile salmonids by double-crested cormorants nesting on East Sand Island in the Columbia River estuary during the 2003-2013 breeding seasons. Estimates are based on fish identified in cormorant foregut samples and bioenergetics calculations.

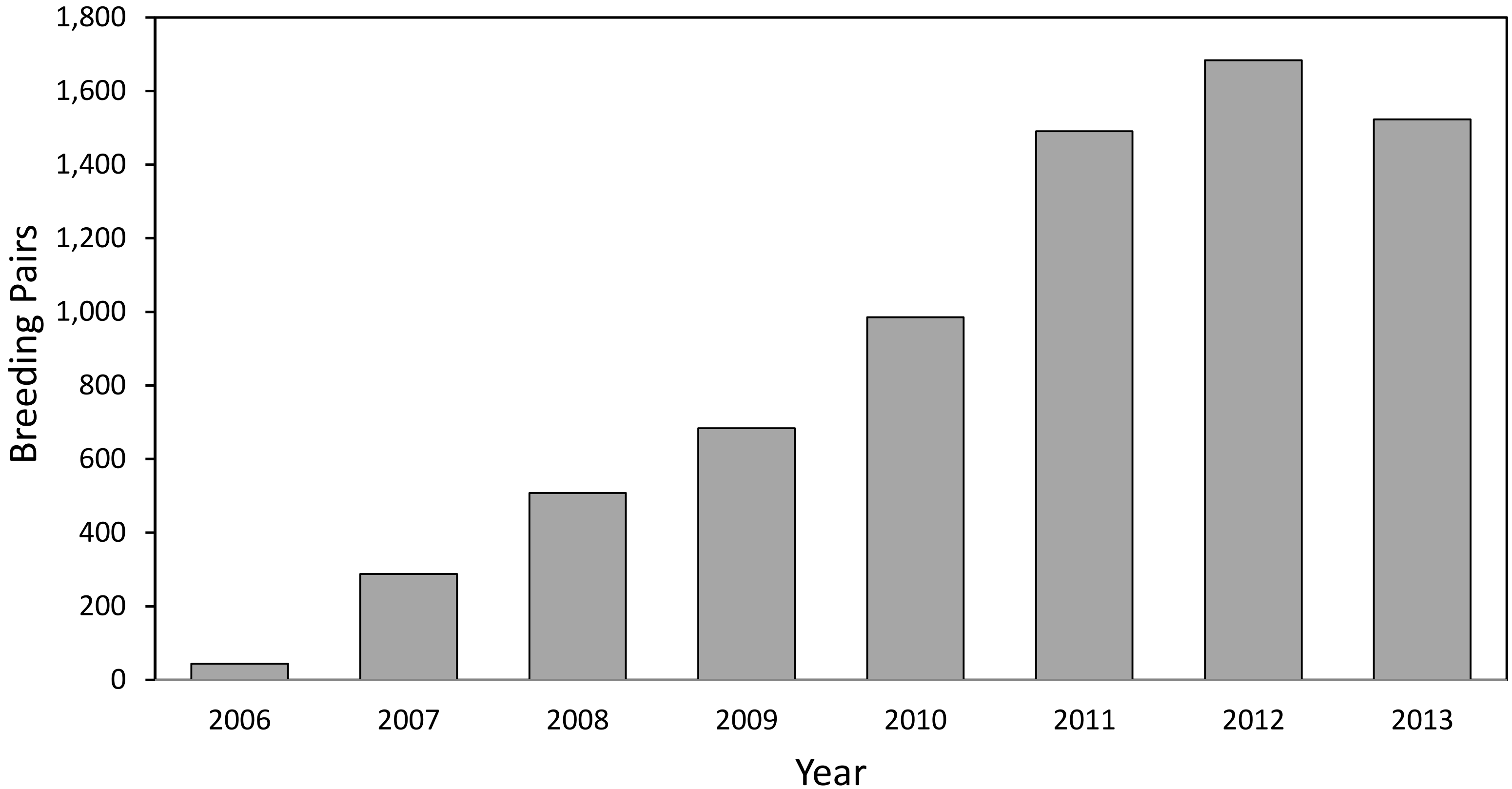


Figure 71. Size of the Brandt's cormorant breeding colony (number of breeding pairs) on East Sand Island in the Columbia River estuary during the 2006-2013 breeding seasons.

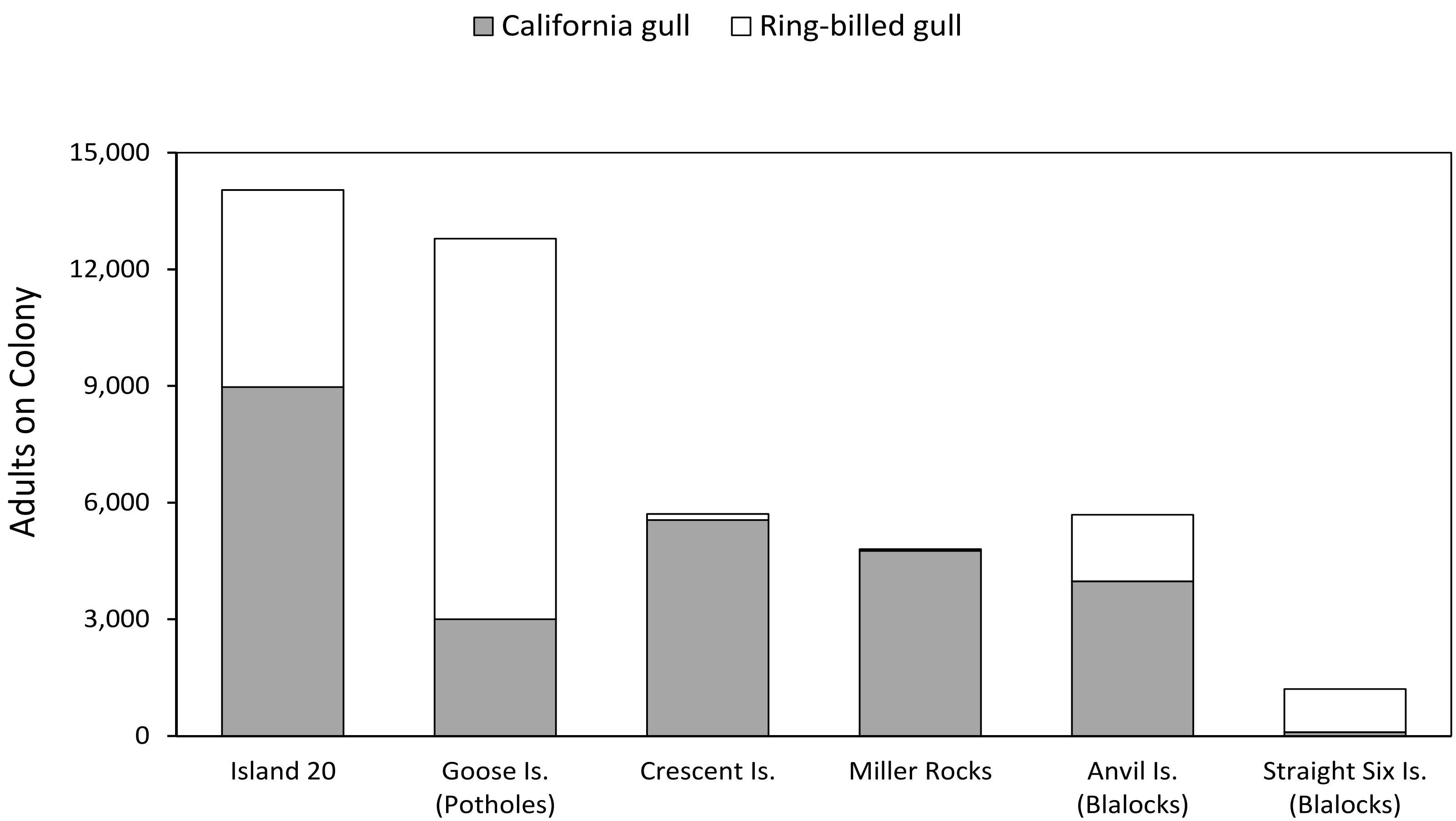


Figure 72. Numbers of adult California and ring-billed **g**ulls counted on aerial photography of six different breeding colonies in the Columbia Plateau region during the 2013 breeding season. Photography was taken late in the incubation period. Three other gull colonies were active in the Columbia Plateau region in 2013, but were not counted: the Harper Island colony in Sprague Lake and the Twinning Island and Goose Island colonies in Banks Lake.

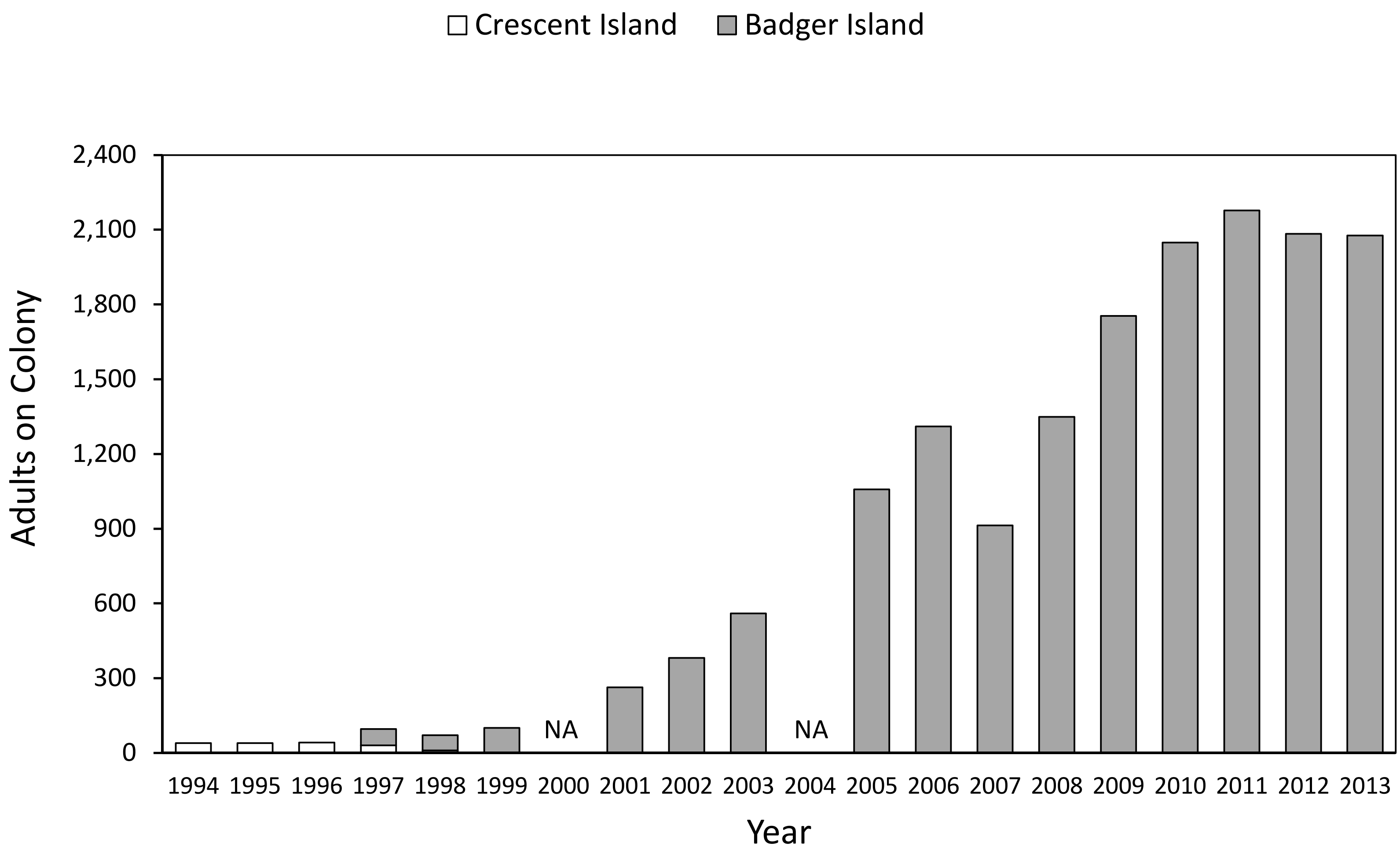


Figure 73. Numbers of adult American white pelicans counted in aerial photography of two colonies on the mid-Columbia River, Badger Island and Crescent Island, during the 1994-2013 breeding seasons. Photography was taken late in the incubation period. Numbers of pelicans on the Badger Island colony were not determined in 2000 and 2004.

Table 1. Caspian tern nesting islands that were built by the U.S. Army Corps of Engineers prior to the 2013 nesting season as part of the federal agencies' Caspian Tern Management Plan for the Columbia River Estuary (USFWS 2005, 2006).

Location	Site	Construction Date	Island Type	Island Size (acres)	Acreage Available in 2013	Notes:
Fern Ridge Reservoir, OR	Fern Ridge	Feb 2008	Rock core	1.0	1.0	Not monitored
Crump Lake, Warner Valley, OR	Crump Lake	Mar 2008	Rock core	1.0	1.0	No social attraction
Summer Lake Wildlife Area, OR	East Link	Jan 2009	Rock core	0.5	0.5	
Summer Lake Wildlife Area, OR	Dutchy Lake	Mar 2009	Floating island	0.5	0.0	Floating island removed
Summer Lake Wildlife Area, OR	Gold Dike	Sep 2009	Rock core	0.5	0.5	
Tule Lake NWR, CA	Sump 1B	Aug 2009	Rock core	2.0	2.0	
Lower Klamath NWR, CA	Orems Unit	Sep 2009	Silt core	1.0	0.0	Land-bridged, Low water
Lower Klamath NWR, CA	Sheepy Lake	Mar 2010	Floating island	0.8	0.8	
Malheur NWR, OR	Malheur Lake	Feb 2012	Rock core	1.0	1.0	
			TOTAL	**8.3**	**6.8**	

Table 2. Number of juvenile salmonid (Chinook, coho, sockeye, and steelhead) PIT tags from 2013 migration year smolts recovered on bird colonies or loafing sites in the Columbia River basin following the 2013 nesting season. Piscivorous waterbird breeding colonies include Brandt's cormorants (BRAC), Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (GULL).

River Segment	Location	Bird Species	Recovered
Estuary	East Sand Is.	CATE	11,830
		DCCO	11,029
		BRAC[1]	477
The Dalles Pool	Miller Rocks	GULL	2,449
John Day Pool	Blalock Islands (Anvil Is.)	GULL	298
		CATE	135
McNary Pool	Crescent Is.	CATE	6,398
		GULL	2,050
	Island 20	GULL	478
Off-river	Goose Is./Potholes Res.	CATE	2,676
Total			37,820

[1] Recoveries on the Brandt's cormorant colony at East Sand Island likely included some tags deposited by double-crested cormorants (see Results).

Table 3. Range of daily detection efficiency estimates for PIT tags sown on bird colonies during the 2013 nesting season. Results were used to adjust predation rate estimates for the number of tags deposited by birds on their nesting colony that were not detected by researchers on the island following the nesting season. Sample sizes and the number of discrete release periods for sown tags are provided (see Section 1.4 for details). Piscivorous waterbird colonies include double-crested cormorants (DCCO), Caspian terns (CATE), and California and ring-billed gulls (GULL).

River Segment	Location	Colony	Sample Size (# release periods)	Date Range	Detection Efficiency
Estuary	East Sand Is.	CATE	300 (3)	3/1 - 8/31	41 - 72%
		DCCO[1]	400 (2)	3/1 - 8/31	62 - 72%
The Dalles Pool	Miller Rocks	GULL	150 (3)	4/1 - 7/31	74 - 90%
John Day Pool	Blalock Islands (Anvil)	GULL	150 (3)	4/1 - 7/31	73 - 90%
McNary Pool	Crescent Is.	CATE	200 (4)	4/1 - 7/31	53 - 90%
		GULL	150 (3)	4/1 - 7/31	54 - 89%
	Island 20	GULL	150 (3)	4/1 - 7/31	65 - 91%
Off-river	Goose Is./Potholes Res.	CATE	400 (4)	4/1 - 7/31	22 - 90%

[1] Values used to calculate predation rates for both double-crested cormorants and Brandt's cormorants (see Methods)

Table 4. On-colony PIT tag deposition rates (DR) used to adjust estimates of 2013 predation rates on juvenile salmonids. Sample size (N) of known consumed PIT-tagged fish used to estimate deposition rates and year when deposition studies were conducted are provided. See Appendix A for description of on-colony deposition rate studies in 2013 and BRNW (2013) for studies conducted in previous years.

Species	Colony	Years	N	DR (95% c.i.)
Caspian tern	East Sand Is., Crescent Is., Goose Is., Blalock Islands	2005-06	362	71% (62-81%)
Double-crested cormorant	East Sand Is.	2013	127	60% (47-73%)
California gull	Miller Rocks	2013	302	20% (15-25%)
	Blalock Islands (Anvil Is.)	2013	302	15% (11-20%)
	Crescent Is.	2013	300	14% (10-20%)
	Island 20	2013	297	15% (10-20%)
Ring-billed gull	California gull deposition rate was applied			
Brandt's cormorant	double-crested cormorant deposition rate was applied			

Table 5. Estimated predation rates (95% confidence interval) on PIT-tagged salmonid smolts last detected at Bonneville Dam on the Columbia River or Sullivan Dam on the Willamette River by avian predators nesting at colonies on East Sand Island in the Columbia River estuary in 2013. Predation rates were adjusted to account for tag loss due to on-colony PIT tag detection efficiency (Table 3) and on-colony PIT tag deposition rate (Table 4). Avian predators include Caspian terns (CATE), double-crested cormorants (DCCO), and Brandt's cormorants (BRAC). The number of PIT-tagged smolts interrogated at Bonneville or Sullivan dams (N) and current U.S. Endangered Species Act (ESA) status of each evolutionarily significant unit (ESU) or distinct population segment (DPS) are provided. Only ESUs/DPSs with > 500 PIT-tagged smolts interrogated passing a dam were evaluated.

			Predation Rate		
ESU/DPS[1]	ESA[2]	N	East Sand Island CATE	East Sand Island DCCO	East Sand Island BRAC[3]
SR Sockeye	E	1,454	0.7% (0.2 - 1.5)	2.6% (1.3 - 4.1)	< 0.1%
SR Spr/Sum Chinook	T	16,167	1.1% (0.8 - 1.5)	2.9% (2.4 - 3.7)	0.2% (0.1 - 0.3)
UCR Spr Chinook	E	3,112	0.6% (0.2 - 1.2)	2.4% (1.6 - 3.5)	< 0.1%
MCR Spr Chinook	NW	4,527	1.3% (0.7 - 1.9)	1.3% (0.8 - 1.9)	0.1% (< 0.1 - 0.3)
SR Fall Chinook	T	4,465	0.8% (0.5 - 1.3)	1.7% (1.1 - 2.4)	0.1% (< 0.1 - 0.2)
UCR Sum/Fall Chinook	NW	4,138	1.4% (0.8 - 2.0)	2.0% (1.3 - 2.8)	0.1% (< 0.1 - 0.2)
UWR Spr Chinook	T	2,629	0.9% (0.4 - 1.4)	0.7% (0.3 - 1.4)	< 0.1%
SR Steelhead	T	8,516	12.5% (10.4 - 15.1)	2.0% (1.5 - 2.7)	0.1% (< 0.1 - 0.2)
UCR Steelhead	T	4,473	8.6% (7.1 - 10.6)	2.7% (1.9 - 3.7)	0.1% (< 0.1 - 0.3)
MCR Steelhead	T	1,865	9.6% (7.1 - 12.5)	1.6% (0.7 - 2.6)	0.1% (< 0.1 - 0.5)

[1] MCR = Middle Columbia River, SR = Snake River, UCR = Upper Columbia River, UWR = Upper Willamette River
[2] E = Endangered, T = Threatened, NW = Not Warranted
[3] Includes some PIT tags deposited by double-crested cormorants (see Results)

Table 6. Estimated predation rates (95% confidence interval) on juvenile salmonids separated by rearing-type (hatchery, wild) and population (ESU/DPS) by Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (GULL) nesting at six colonies in the Columbia River basin during 2013. Predation rates were adjusted for tag loss due to on-colony PIT tag detection efficiency (Table 3) and on-colony deposition rate (Table 4). Predation rates are based on numbers of PIT-tagged fish, per ESU/DPS, last interrogated passing dams on the Columbia or Snake rivers. Dams include Bonneville Dam (BON), McNary Dam (MCN), Lower Monumental Dam (LMO), and Rock Island Dam (RIS). Only ESUs/DPSs with > 500 PIT-tagged hatchery and wild smolts, per interrogation site, were evaluated. Only colonies with predation rates > 1.0% (see Tables 5, 6, and 9), those deemed large enough to compare with biologically meaningful differences, were evaluated.

Island	Bird Species	Salmonid ESU/DPS	Interrogation Site	Predation Rate	
				Hatchery	Wild
East Sand Is.	CATE	SR steelhead	BON	12.5% (10.4-15.3)	12.4% (9.8-15.6)
		UCR steelhead		8.7% (7.0-10.8)	7.5% (3.7-12.2)
		SR Spr. Chinook		1.2% (0.9-1.6)	0.8% (0.2-1.5)
		UCR Spr. Chinook		0.7% (0.2-1.2)	0.5% (<0.1-1.6)
	DCCO	SR steelhead	BON	2.0% (1.4-2.7)	2.0% (1.0-3.0)
		UCR steelhead		2.5% (1.7-3.5)	4.6% (1.7-8.3)
		SR Spr. Chinook		3.1% (2.5-3.9)	2.1% (1.1-3.2)
		UCR Spr. Chinook		2.6% (1.6-3.7)	1.6% (<0.1-3.8)
Miller Rocks	GULL	SR steelhead	MCN	5.0% (3.5-7.0)	4.0% (2.1-6.5)
		UCR steelhead		9.5% (6.1-14.2)	4.7% (1.0-10.1)
		UCR Sp. Chinook		1.4% (0.6-2.3)	1.2% (<0.1-2.9)
Blalock Islands	GULL	SR steelhead	MCN	1.1% (0.4-2.0)	0.7% (<0.1-1.8)
Crescent Is.	CATE	SR steelhead	LMN or RIS	3.0% (2.4-3.6)	2.6% (1.9-3.3)
		UCR steelhead		2.9% (2.2-3.7)	2.5% (1.6-3.7)
	GULL	SR steelhead	LMN or RIS	6.6% (4.3-10.3)	1.8% (0.6-3.4)
		UCR steelhead		7.6% (4.7-12.0)	2.4% (0.5-5.3)
Island 20	GULL	UCR steelhead	RIS	1.8% (0.7-3.4)	<0.1%
Goose Is. /Potholes Res.	CATE	UCR steelhead	RIS	17.2% (14.5-20.5)	8.5% (6.0-11.4)

Table 7. Estimated predation rates (95% confidence interval) on PIT-tagged salmonid smolts last detected at Lower Monumental Dam on the Snake River or Rock Island Dam on the upper Columbia River by avian predators nesting at colonies on Crescent Island, Island 20, or Goose Island/Potholes Reservoir during 2013. Predation rates were adjusted to account for tag loss due to on-colony PIT tag detection efficiency (Table 3) and on-colony PIT tag deposition rate (Table 4). Colonies include those of Caspian terns (CATE) and California and ring-billed gulls (GULLS). The number of PIT-tagged smolts interrogated at Lower Monumental or Rock Island dams (N) and current U.S. Endangered Species Act (ESA) status of each evolutionarily significant unit (ESU) or distinct population segment (DPS) are provided. Only ESUs/DPSs with > 500 PIT-tagged smolts that were interrogated while passing a dam were evaluated.

			Predation Rate			
ESU[1]	ESA[2]	N	Crescent Is. CATE	Crescent Is. GULLS	Island 20 GULLS	Goose Is. CATE
SR Sockeye	E	1,697	0.5% (0.1 - 1.0)	1.2% (0.1 - 3.1)	< 0.1%	< 0.1%
SR Spr/Sum Chinook	T	21,611	0.5% (0.4 - 0.7)	0.7% (0.3 - 1.2)	0.2% (< 0.1 - 0.4)	< 0.1%
UCR Spr Chinook	E	992	0.2% (< 0.1 - 0.6)	< 0.1%	< 0.1%	2.1% (0.7 - 4.0)
SR Fall Chinook	T	6,423	0.6% (0.4 - 0.9)	0.5% (< 0.1 - 1.3)	< 0.1%	< 0.1%
UCR Sum/Fall Chinook	NW	2,907	< 0.1%	< 0.1%	< 0.1%	0.3% (0.1 - 0.6)
SR Steelhead	T	11,958	2.8% (2.4 - 3.4)	4.8% (3.2 - 7.2)	0.5% (0.2 - 1.0)	0.1% (< 0.1 - 0.2)
UCR Steelhead	T	5,893	2.8% (2.2 - 3.5)	6.1% (3.9 - 9.6)	1.3% (0.5 - 2.4)	14.9% (12.7 - 17.8)

[1] SR = Snake River, UCR = Upper Columbia River
[2] E = Endangered, T = Threatened, NW = Not Warranted

Table 8. Cumulative estimated predation rates (95% confidence interval) on steelhead smolts PIT-tagged and released at Rock Island Dam by piscivorous waterbirds nesting at colonies in the Columbia River basin during 2008 - 2013. Predation rates were adjusted to account for tag loss due to on-colony PIT tag detection efficiency (Table 3; see BNRW 2013 for previous years) and on-colony PIT tag deposition rate (Table 4, see BRNW 2009-2013 for previous years). Annual predation rates were calculated for the number of PIT-tagged steelhead smolts released (n) at Rock Island Dam, but were not adjusted for steelhead survival to the vicinity of the bird colony (river kilometer [RKM]). Dashes indicate the colony was not scanned in that year.

				Predation rates				
Location	Bird species[1]	RKM	2008 (n = 7,266)	2009 (n = 7,109)	2010 (n = 7,364)	2011 (n = 7,756)	2012 (n = 6,712)	2013 (n = 5,893)
Banks Lake	CATE	Off-river	0.1% [3]	0.1% (< 0.1 - 0.2)	0.1% (< 0.1 - 0.3)	-	0.1% (< 0.1 - 0.3)	-
Harper Is.	CATE	Off-river	-	-	-	-	< 0.1%	-
	DCCO	Off-river	-	-	-	-	< 0.1%	-
N. Potholes	DCCO	Off-river	-	-	-	-	0.3% (< 0.1 - 0.8)	-
Goose Is.	CATE	Off-river	10.8% (9.4 - 12.5)	21.9% (18.7 - 25.6)	13.5% (11.5 - 16.1)	12.7% (10.7 - 15.2)	17.0% (13.7 - 21.5)	14.9% (12.7-17.8)
	GULLS	Off-river	-	-	-	-	2.8% (1.2 - 6.1)	-
Island 20	GULLS		-	-	-	-	-	1.2% (0.4 - 2.1)
Foundation Is.	DCCO	518	0.3% (0.1 - 0.6)	0.3% (0.1 - 0.5)	0.1% (< 0.1 - 0.4)	0.3% (0.1 - 0.6)	0.5% (0.1 - 0.9)	-
Badger Is.	AWPE[2]	512	0.1% (< 0.1 - 0.2)	0.3% (0.1 - 0.4)	0.1% (< 0.1 - 0.2)	0.1% (< 0.1 - 0.1)	0.1% (< 0.1 - 0.2)	-
	CATE	512	-	-	-	0.7% (0.2 - 1.4)	0.2% (< 0.1 - 0.7)	-
Crescent Is.	CATE	510	2.9% (2.3 - 3.6)	2.2% (1.7-2.7)	1.7% (1.3-2.2)	2.4% (1.9 - 2.9)	1.2% (0.8 -1.7)	2.8% (2.2 - 3.5)
	GULLS	510	2.2% (1.3 - 3.3)	5.6% (3.9-7.7)	6.2% (4.5-8.4)	3.1% (1.9-4.7)	3.9% (2.1-5.7)	5.6% (3.4 - 8.7)
Blalock Islands	CATE	441	0.5% (0.3 - 0.7)	0.2% (0.1-0.4)	0.4% (0.3-0.6)	<0.1%	-	< 0.1%
	GULLS	441	-	-	-	-	-	1.7% (0.8 - 2.9)
Miller Rocks	GULLS	331	4.1% (2.8 - 5.8)	4.3% (2.9-6.2)	3.6% (2.5-5.3)	2.8% (1.8-4.1)	2.7% (1.6-4.0)	3.0% (1.9 - 4.4)
Miller Sands Spit	AWPE[2]	38	-	-	-	<0.1%	-	-
East Sand Is.	CATE	8	9.0% (8.0 - 10.0)	8.3% (7.3 - 9.4)	7.6% (6.6 - 8.7)	3.9% (3.3 - 4.7)	3.3% (2.6 - 4.1)	6.0% (4.8 - 7.5)
East Sand Is.	BRAC	8	-	< 0.1%	< 0.1%	0.2% (< 0.1 - 0.4)	0.1% (< 0.1 - 0.2)	0.1% (< 0.1 - 0.2)
East Sand Is.	DCCO	8	3.2% (2.4 - 4.3)	2.9% (2.0 - 3.8)	3.5% (2.6 - 4.5)	4.4% (3.4 - 5.5)	3.7% (2.8 - 4.9)	1.2% (0.8 - 1.7)
Total			33.6% (30.9-36.8)	47.3% (43.1 - 52.1)	37.5% (34.4 - 41.7)	31.0% (28.0 - 34.7)	36.4% (31.7 - 42.6)	36.3% (32.6-41.3)

[1] CATE = Caspian tern; DCCO = double-crested cormorant; BRAC = Brandt's cormorant; GULLS = ring-billed and California gulls; AWPE = American white pelican

[2] Predation rates by American white pelicans were not adjusted for on-colony deposition rate due to lack of empirical data and should be considered minimum estimates.

[3] Confidence interval unstable due to high degree of uncertainty in on-colony detection efficiency estimates during 2008

Table 9. Estimated predation rates (95% confidence interval) on PIT-tagged salmonid smolts last detected at McNary Dam on the Columbia River by Caspian terns (CATE) nesting on the Blalock Islands, California and ring-billed gulls (GULLS) nesting on the Blalock Islands, and California gulls nesting Miller Rocks in 2013. Predation rates were adjusted to account for tag loss due to on-colony PIT tag detection efficiency (Table 3) and on-colony PIT tag deposition rate (Table 4). The number of PIT-tagged smolts interrogated at McNary Dam (N) and current U.S. Endangered Species Act (ESA) status of each evolutionarily significant unit (ESU) or distinct population segment (DPS) are provided. Only ESUs with > 500 PIT-tagged smolts that were interrogated while passing a dam were evaluated.

			Predation Rates		
ESU[1]	ESA[2]	N	Blalock Is. CATE	Blalock Is. GULLS	Miller Rocks GULLS
SR Sockeye	E	1,213	< 0.1%	0.7% (0.1 - 2.3)	4.5% (1.8 - 8.1)
SR Spr/Sum Chinook	T	47,685	< 0.1%	< 0.1%	0.8% (0.6 - 1.2)
UCR Spr Chinook	E	6,778	< 0.1%	< 0.1%	1.3% (0.7 - 2.2)
SR Fall Chinook	T	14,398	< 0.1%	0.2% (< 0.1 - 0.5)	1.8% (1.2 - 2.6)
UCR Sum/Fall Chinook	NW	13,113	< 0.1%	0.4% (0.1 - 0.8)	1.3% (0.8 - 1.9)
SR Steelhead	T	9,391	0.1% (< 0.1-0.2)	1.1% (0.5 - 1.9)	4.7% (3.4 - 6.6)
UCR Steelhead	T	2,621	< 0.1%	0.6% (0 - 1.7)	8.5% (5.6 - 12.3)

[1] SR = Snake River, UCR = Upper Columbia River
[2] E = Endangered, T = Threatened, NW = Not Warranted

APPENDIX A: PIT Tag Deposition Studies

Presented here are methods and results of PIT tag deposition studies conducted at selected piscivorous waterbird colonies where estimates of tag loss due to off-colony PIT tag deposition and/or PIT tag damage (collectively referred to as “PIT tag deposition”) are needed to calculate actual or best estimates of predation rates on salmonid smolts in 2013 (see Section 1.4 for predation rate calculation methods). Studies of PIT tag deposition rates were conducted for double-crested cormorants (*Phalacrocorax auritus*; hereafter “cormorants”) nesting on East Sand Island in the Columbia River estuary and at four California gull (*Larus californicus*; hereafter “gull”) colonies in the Columbia Plateau region. Caspian tern (*Hydroprogne caspia)* deposition rate studies were not conducted in 2013 (See BRNW 2013 for methods and results of tern deposition rate studies).

In 2013, we also conducted an independent experiment to investigate possible causes of low PIT tag deposition rates that were observed for gulls in 2012 (BRNW 2013). For example, on-colony gull deposition rates averaged just 17% in 2012, one of the lowest estimates of deposition rates of any piscivorous waterbird species ever reported (BRNW 2013). The low estimate suggested that the vast majority of PIT tags consumed by gulls were either deposited at off-colony loafing or staging areas, or tags were somehow damaged (e.g., broken) during passage through the gastrointestinal (GI) tract of the gull. Unlike terns and cormorants, gulls macerate their food in their gizzard during digestion, and other researchers have found crushed PIT tags in the foreguts of California gulls (Zorich et al. 2011). An independent study quantifying PIT tag damage rates during passage through the GI tract of gulls was conducted to better understand which mechanism, off-colony deposition or tag damage during digestion, was responsible for low PIT tag deposition rates by gulls.

STUDY AREA

PIT tag deposition rates were evaluated in three different species of piscivorous waterbirds nesting at seven different colonies located in the Columbia River basin: (1) cormorants nesting on East Sand Island (ESI) in the Columbia River estuary, (2) gulls nesting on Miller Rocks (MRO) in The Dalles Pool, (3) gulls nesting on the Blalock Islands (BLI) in John Day Pool, (4) gulls nesting on Crescent Island (CSI) in McNary Pool, (5) gulls nesting on Island 20 (I20) in McNary Pool, (6) terns nesting on East Sand Island in the Columbia River estuary, and (7) terns nesting on Crescent Island in McNary Pool (Map 1). Deposition rate data from cormorants and gulls were collected in 2012-2013 and data from 2013 presented here; data from terns were collected in 2005-2006 and are presented in BRNW 2013.

METHODS

Fish Consumption

Double-crested Cormorants

During 2013, the cormorant colony at ESI was monitored at least once per week to document nesting chronology. Stages of the nesting cycle were categorized as (1) nest-building, (2) egg incubation, or (3) chick-rearing, depending on the nesting stage of the majority of breeding cormorants observed during each weekly visit. Hatchery rainbow trout (fork length = 80-220 mm) were euthanized, PIT-tagged (12 mm, 134.2 kHz full-duplex), and thrown to cormorants nesting adjacent to on-colony observation blinds during each of the stages of the nesting cycle. To account for possible circadian patterns in deposition rates, approximately equal numbers of PIT-tagged trout were fed to cormorants in the mornings (8:00 – 11:00) and evenings (15:00 – 18:00 PDT). Trout were thrown 1 to 5 meters from each blind in various directions, an area encompassing the nests of approximately 250 - 300 breeding pairs of cormorants. Only trout consumed by nesting adult cormorants were included in the study (n = 37 - 47 per feeding period, for a total of 127 PIT-tagged trout consumed by cormorants; Table A1).

California Gulls

During 2013, gull colonies at Miller Rocks (MRO), the Blalock Islands (BLI), Crescent Island (CSI), and Island 20 (I20) were monitored once per week to determine nesting chronology. Stages of the nesting cycle were categorized using the same methods as those presented above for cormorants, and roughly equal numbers of PIT-tagged hatchery trout (fork length = 80 – 220 mm) were thrown to adult birds during three discrete nesting stages. Similar to cormorants, to account for possible circadian patterns in deposition rates, approximately equal numbers of PIT-tagged trout were fed to gulls in the mornings (8:00 – 11:00) and the evenings (15:00 - 18:00). Also similar to cormorants, trout were thrown to gulls from an on-colony observation blind to minimize disturbance to nesting birds. Trout were thrown 3 to 20 meters from the blind in all directions, an area encompassing approximately 200-250 nesting pairs of gulls at each island. Only trout consumed by nesting adult gulls were included in the study (n = 98 - 103 consumed trout colony^{-1} period^{-1}, for a total of 1,201 PIT-tagged trout consumed by gulls; Table A2).

To determine which mechanism, off-colony PIT tag deposition or PIT tag damage during digestion, was responsible for low on-colony PIT tag deposition rates in gulls, a different group of nesting gulls on MRO was fed trout containing two different tag types, with both tag types placed in each trout (double-tagged fish): (1) a regular or standard glass PIT tag and (2) a standard glass PIT tag encapsulated in polyolefin tubing. Polyolefin tubing is a plastic wrap that adheres to glass to protect it from environmental damage, thereby making the tag much less likely to break during digestion. If damage to standard PIT tags is the primary reason for low on-colony deposition rates, we theorized that a larger proportion of polyolefin encapsulated tags would be deposited on-colony compared to standard tags. To account for possible differences in foregut contents of gulls (e.g., stones, fish bones, cherry pits, or other solid materials that could damage tags during digestion), we double-tagged each fish to ensure that each tag type (standard, encapsulated) experienced the same foregut conditions during digestion.

PIT Tag Recovery

Recovery methods for PIT tags on bird colonies are the same as those described in Section 1.4, and are only briefly summarized here. PIT tags were recovered from each bird colony once nesting birds had dispersed after the nesting season (August to November). PIT tag antennas were used to detect PIT tags *in situ* by systematically scanning the area that was occupied by breeding birds during the nesting season (Evans et al. 2012; Zamon et al. 2013). The area occupied by birds on each colony was determined from aerial photography of the colony and visits to the colony during the nesting season.

Detection Efficiency

Methods to measure PIT tag detection efficiency are the same as those described in Section 1.4, and are only briefly summarized here. PIT tags with known tag codes were intentionally sown on each bird colony (hereafter referred to as "control tags") throughout the nesting season to quantify on-colony PIT tag detection efficiency. Control tags were the same dimensions and weight as PIT tags used to mark trout in this study (12 mm, 134.2 kHz full-duplex). The total number of control tags sown varied by colony, with sample sizes ranging from 150 to 400 control tags colony^{-1} year^{-1}. The number of discrete time periods when control tags were sown also varied due to limited accessibility to some bird colonies, but was no less than twice (at the beginning and at the end of the nesting season) and no more than three times. Logistic regression was used model detection efficiency as a function of time at each colony (see equation below). Results of detection efficiency trails at each cormorant and gull colony are presented in Table 3 and Appendix D, Figure D2.

To account for possible differences in detection efficiency between standard glass PIT tags and polyolefin encapsulated PIT tags - those used to quantify PIT tag damage by gulls during digestion - equal numbers of both tag types (standard, encapsulated) were sown on the MRO gull colony. Results of detection efficiency trails for encapsulated tags are presented below and in Appendix D, Figure D2.

Deposition rate estimates (see below) for each colony and PIT tag type (standard, encapsulated) were then corrected for colony-specific detection efficiencies.

Deposition Rate Estimation

Deposition rates of PIT tags from trout consumed by nesting adult birds were calculated using an iterative multi-step approach. First, logistic regression was used to interpolate colony-specific detection efficiencies for the time periods of interest, whereby a binary response of control tag detections (detected or not detected) was modeled as a function of time since control tags were placed on the bird colony (eq. 1).

$$(1) \quad \hat{p_j} = \frac{e^{(\beta_0 + \beta_1 t_j)}}{1 + e^{(\beta_0 + \beta_1 t_j)}},$$

where $\hat{p}_j$ is the probability of detecting a tag deposited on day *j*, β_0 is the regression intercept, β_1 is the regression slope, and t_j is the independent variable for date *j*. The estimated number of trout PIT tags consumed on day *j* that were deposited on-colony ($\hat{d_j}$) is the number of recovered trout tags consumed on day *j* (r_j), divided by the probability of detecting a tag deposited on-colony on day *j* ($\hat{p_j}$) (eq. 2)

$$(2) \quad \hat{d_j} = {}^{r_j}\!/_{\hat{p_j}}$$

Annual deposition rates ($\hat{\phi}$) were estimated by dividing the estimated number of trout tags deposited on-colony ($\hat{d_j}$) by the sum of trout tags consumed (a_j) across the entire season. Confidence intervals for deposition rates were estimated by a bootstrapping simulation technique (Efron & Tibshirani 1986; Manly 1998). The bootstrapping analysis consisted of 2,000 iterations of all calculations, with each iteration representing a unique bootstrap re-sample (random sample with replacement) of all datasets. The 2.5th and 97.5th quartiles were used to estimate the limits of a 95% confidence interval for deposition rate. Deposition rates were considered significantly different if there was no overlap between 95% confidence intervals.

RESULTS

Deposition Rates

Double-crested Cormorants

Adult cormorants nesting at ESI consumed 127 PIT-tagged trout during volitional feeding studies in 2013 (Table A1). Although small sample sizes limit the statistical robustness of comparisons, there were no significant differences in deposition rates as a function of nesting stage or time of day when a PIT-tagged trout was consumed by a nesting cormorant (Table A1). When combined (all periods), annual PIT tag deposition rate for cormorants was estimated at 60% (95% c.i. = 47 - 73; Table A1) in 2013.

There was evidence that the estimate of cormorant deposition rate at East Sand Island in 2013 (60%; 95% c.i. = 47 - 73; Table A1) was higher than that observed in 2012 (44%; 95% c.i. = 36 - 51; BRNW 2013), but small sample size in 2013 limited the robustness of statistical comparisons between years. Combining data from cormorant studies of PIT tag deposition rates in 2012 and 2013 yields an estimate of on-colony deposition rate of 48% (95% c.i. = 42 - 55).

The sample size goal for volitional feedings studies at the ESI cormorant colony in 2013 was 300 consumed PIT-tagged trout or 100 per stage of the nesting cycle. Unfortunately, unlike 2012, when sample size goals were achieved (BRNW 2013), cormorants were reluctant to consume PIT-tagged trout tossed from observations blinds in 2013. As a result, despite presenting well over 600 PIT-tagged trout to cormorants during the 2013 nesting season, only 127 were consumed by cormorants near observation blinds.

California Gull

Adult gulls consumed 1,201 PIT-tagged trout during volitional feeding studies at MRO (n = 302), BLI (n = 302), CSI (n = 300), and I20 (n = 297) in 2013 (Table A2). There was no significant difference in annual deposition rates by colony site, with annual estimates 20% (95% c.i. = 15 - 25%) at MRO, 15% (95% c.i. = 11 - 20%) at BLI, 14% (95% c.i. = 10 -2 0%) at CSI, and 15% (95% c.i. = 10 - 20%) at I20 (Table A2). There was some evidence that deposition rates for gulls at a given colony varied across the nesting season, with deposition rates during the chick-rearing period often, but not always, significantly higher than those during the nest-building or egg-incubation periods (Table A2). There was no evidence, however, that deposition rates varied by the time of day (morning, evening) when a PIT-tagged fish was consumed by a nesting gull, with no significant difference at all four gull colonies (Table A2). Combining data from all four gull colonies resulted in an overall on-colony deposition rate of 16% (95% c.i. = 14 - 19%) in 2013 (Table A2).

Deposition rate estimates at gull colonies in 2013 (16%; 95% c.i. = 14 - 19%; Table A2) were remarkably similar to those in 2012 (17%; 95% c.i. = 13 - 21%; BRNW 2013), with no significant difference in annual estimates observed by year or by colony. When all colonies, periods, and years are combined, a single estimate of on-colony deposition rates for gulls was 17% (95% c.i. = 14 - 19%).

PIT Tag Digestion Damage - Adult gulls consumed 300 double-tagged (1 standard tag plus 1 encapsulated tag) trout at MRO during 2013, with equal numbers consumed during each of the three stages of the nesting cycle. There were significant differences in deposition rates between PIT tag types during each stage of the nesting cycle: on-colony deposition rates of encapsulated PIT tags were 3 to 4 times higher than those of standard glass PIT tags (Figure A1). The annual deposition rate for encapsulated tags was 57% (95% c.i. = 50 - 65%), compared to just 19% (95% c.i. = 14 - 24%) for standard PIT tags implanted in the same fish (Figure A1). Similar to the standard PIT tags, there was some evidence that deposition rates for encapsulated tags varied across the nesting season, with higher deposition rates recorded during the nesting-building and chick-rearing periods (Figure A1).

DISCUSSION

A technique to measure PIT tag deposition rates in Caspian terns, double-crested cormorants, and California gulls was developed and used to generate more accurate estimates of predation rates on PIT-tagged smolts for these species of fish-eating waterbirds. Our results demonstrated substantial differences in PIT tag deposition rates among species of avian predators, with annual deposition rates significantly higher for Caspian terns, followed by double-crested cormorants, and lowest for California gulls (Figure A2). Species-specific deposition rates can be influenced by a number factors, including differences in the rate at which PIT tags are damaged during digestion, or rates of deposition at off-colony areas utilized by birds during the breeding season. For instance, the diet of Caspian terns and double-crested cormorants is nearly entirely fish (Collis et al. 2002), while the diet of gulls is much more diverse and often includes items that could damage PIT tags during digestion (e.g., small stones, cherry pits, and other hard objects that help macerate food during digestion (Winkler 1996; Collis et al. 2002). The finding that encapsulated tags were 3 to 4 times more likely to be recovered on a gull colony compared to a standard glass PIT tags supports the hypothesis that the hard materials ingested by gulls are damaging PIT tags during residence time in the gut and provides strong evidence that on-colony egestion of unreadable PIT tags, as opposed to off-colony deposition, is a major factor causing the low on-colony deposition rates of functional smolt PIT tags by California gulls. The incidence of PIT tag damage in Caspian tern and double-crested cormorant GI tracts is, however, unknown, but is presumably much less than for gulls given that annual on-colony deposition rates were nearly 50% and 70% in cormorants and terns, respectively.

Despite significant differences in PIT tag deposition rates between bird species (terns, cormorants, gulls), deposition rates were relatively consistent within the same species. This was especially true for California gull colonies, where estimates of deposition rates were remarkably similar among colonies and between years. Some variation in deposition rates as a function of stage of the nesting cycle was observed for both cormorant and gull colonies; however, these trends were often inconsistent and the overall seasonal difference in deposition rates was generally less than 10%. Taken as a whole, our results suggest that annual, species-specific estimates of PIT tag deposition rates can be applied be applied with some confidence across colonies and years.

Birds are often top predators in aquatic ecosystems (Steinmetz et al. 2003); however, fish tag recovery studies often produce only minimum estimates of avian predation rates on fish populations (Collis et al. 2001; Ryan et al. 2001; Evans et al. 2011; Evans et al. 2012; Frechette et al. 2012). Detections of fish tags on bird colonies have helped identify avian predation as a substantial source of fish mortality in multiple freshwater and estuarine ecosystems (Collis et al. 2001; Evans et al. 2011; Evans et al. 2012; Frechette et al. 2012). In most of these studies however, predation rates were considered minimum estimates due to a lack of measurements of the proportion of ingested fish tags that are deposited in a detectable condition on bird colony. This study provides estimates of PIT tag deposition rates for three species of fish-eating birds, and suggests methods for incorporating those rates into estimates of predation rates. Incorporation of on-colony tag deposition rates into models for estimating predation rates increased avian predation rates by roughly a factor of 1.5 for Caspian terns, 2.0 for double-crested cormorants, and 6.0 for California gulls, compared to estimates that do not account for the proportion of ingested PIT tags that are egested on the breeding colony in workable condition.

Previous studies have highlighted several challenges with interpreting fish tag recoveries on bird breeding colonies (Collis et al. 2001; Ryan et al. 2001; Evans et al. 2011; Evans et al. 2012; Frechette et al. 2012). First, a tag may be deposited on-colony and not found (detection efficiency). Second, a fish may be consumed, but the tag may not be deposited on-colony in working order (on-colony deposition rate). PIT tag deposition rates for California gulls were significantly lower than those for Caspian terns and double-crested cormorants. Low on-colony deposition rates raise the possibility that recovery of just a few fish tags could nevertheless be associated with significant impacts to fish survival rates. Low on-colony deposition rates increase the uncertainty of the magnitude of impacts from gull predation associated with various colonies in the Columbia River basin. Field methods to measure on-colony deposition rates and analytical methods that best integrate this uncertainty are required to provide accurate estimates of predation rates by gulls for on-going management and monitoring activities.

Studies are needed to quantify on-colony PIT tag deposition rates for other species fish-eating birds in order to improve estimates of predation rates on fish species of conservation concern. Results of this study prove that unaccounted for variation in on-colony PIT tag deposition rates may lead to large differences in the interpretation of data on on-colony PIT tag recoveries. The methods described here could be used to investigate other bird species on whose breeding colonies recoveries of fish PIT tags have been documented, including but not limited to: American white pelicans (*Pelecanus erythrorhynchos*; predation rates presented in Evans et al. 2012), Brandt's cormorants (*Phalacrocorax penicillatus*; Frechette et al. 2012), great cormorants (*Phalacrocorax carbo*; Koed et al. 2006), northern gannets (*Sula bassanus*; Montevecchi et al. 1988), ring-billed gulls (*Larus delawarensis*; Evans et al. 2012), and western gulls (*Larus occidentalis*; Frechette et al. 2012). Similarly, methods used in this study could be replicated to estimate on-colony deposition rates for other fish tags, such as acoustic tags (used to evaluate avian predation by Halfyard et al. [2012]), Carlin tags (Boström et al. 2009), coded wire tags (Lovvorn et al. 1999; Evans et al. 2011), and radio tags (Jepsen et al. 1998; Kaeding 2002; Schreck et al. 2006). As demonstrated by the use of encapsulated PIT tags in California gulls, the type of tag used to estimate predation rates may be important because deposition rates can be tag-specific.

Predation is a key ecological process influencing fish populations (Sih 1987). Impacts of avian predation on fish populations can be quantified using mark-recovery techniques uniquely adapted to specific obstacles within a given system. Incorporation of on-colony tag deposition rates, however, is required to properly estimate avian predation rates and their associated impacts on fish survival when recovering tags from bird colonies. Accurately quantifying causes of fish mortality will aid in the development of management strategies that contribute to the understanding of important population parameters and management plans to recover of ESA-listed fish populations.

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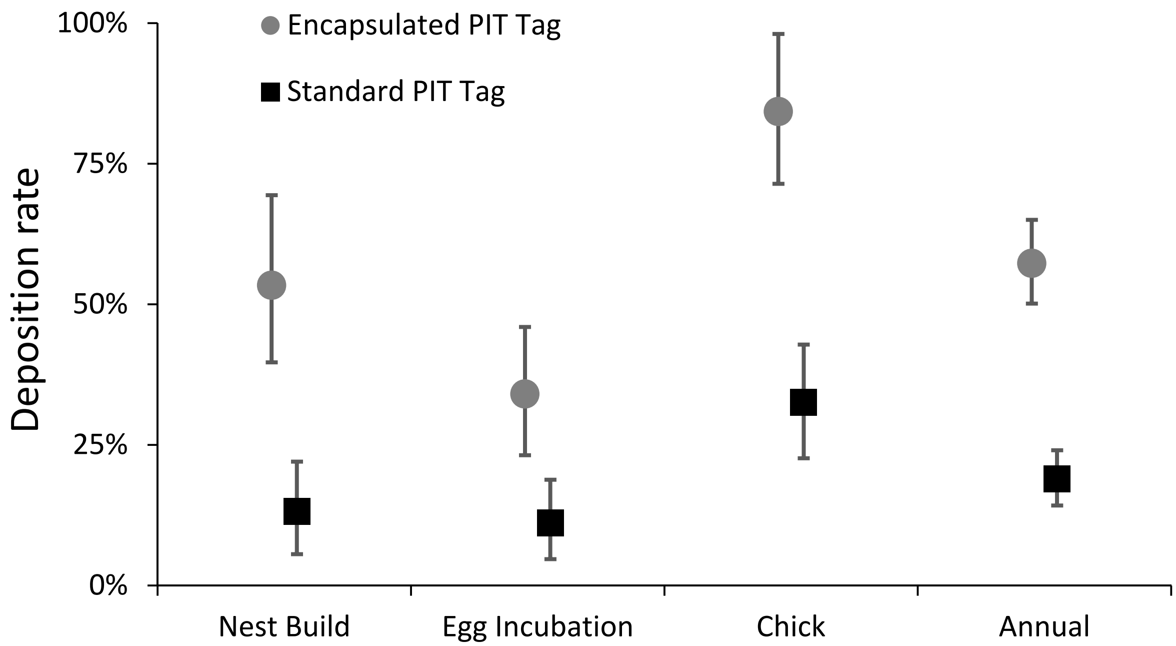

Figure A1. Estimated on-colony PIT tag deposition rates for standard glass PIT tags and PIT tags encapsulated in polyolefin by California gulls nesting at Miller Rocks in 2013. Estimates are provided by study period (nest-building, egg-incubation, and chick-rearing; see Methods). Error bars represent 95% confidence intervals.

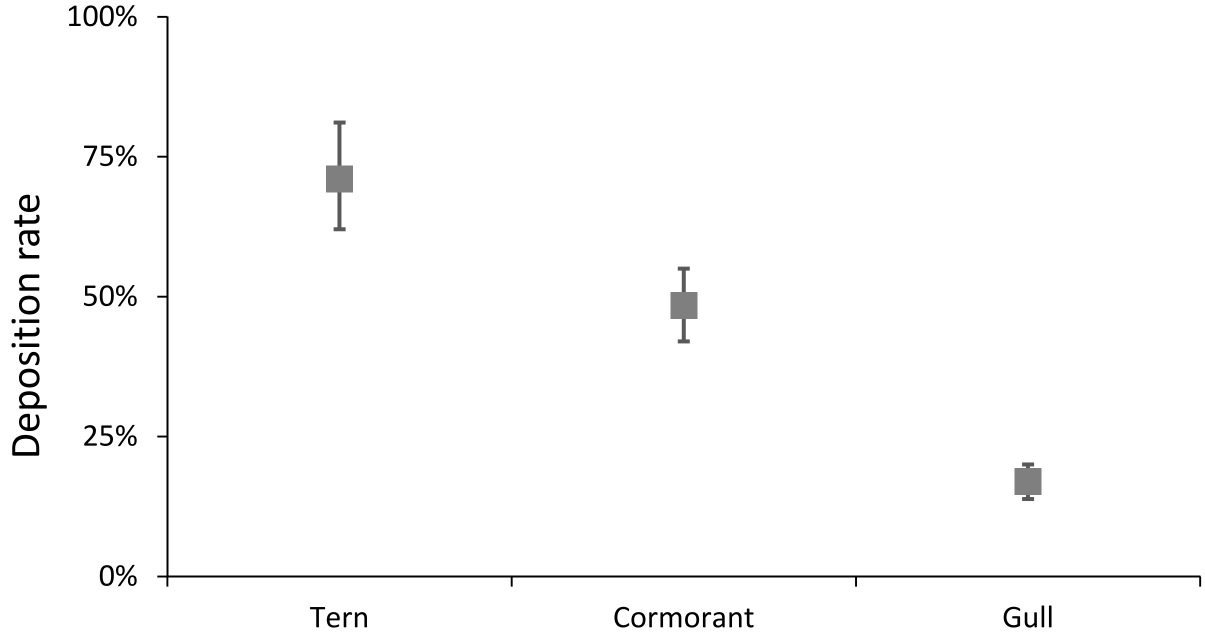

Figure A2. Annual average estimates of on-colony PIT tag deposition rates for Caspian terns (tern), double-crested cormorants (cormorant), and California gulls (gull). Tern data were collected during 2005 and 2006 (BRNW 2013), while cormorant and gull data were collected during 2012 (BRNW 2013) and 2013. Error bars represent 95% confidence intervals.

Table A1. On-colony PIT tag deposition rates for double-crested cormorant nesting at East Sand Island in 2013. Experiments were conducted by feeding PIT-tagged trout to actively-nesting adult cormorants near observations blinds (Volitional). PIT tags recovered (Recovered) were adjusted for date and colony-specific detection efficiency (DE) to estimate the total number of tags deposited on the island (Deposited) and the deposition rate.

Location	Method	Year	Period	Date	Eaten	Recovered	DE	Deposited	Deposition Rate (95% c.i.)
ESI	Volitional	2013	Nest building	5/1	37	16	0.66	24	66% (41-91%)
ESI	Volitional	2013	Incubation	5/27	47	23	0.67	34	73% (51-96%)
ESI	Volitional	2013	Chick	6/24	43	12	0.68	18	41% (23-62%)
ESI	Volitional	2013	AM	-	68	25	-	37	55% (39-73%)
ESI	Volitional	2013	PM	-	59	26	-	39	66% (47-85%)
ESI	Volitional	2013	Annual	-	127	51	-	76	60% (47-73%)

Table A2. On-colony PIT tag deposition rates for California gulls nesting at Miller Rocks, the Blalock Islands, Crescent Island, and Island 20 in 2013. Experiments were conducted by feeding PIT-tagged trout to actively-nesting gulls (Volitional). PIT tags recovered (Recovered) were adjusted for date and colony-specific detection efficiency (DE) to estimate the total number of tags deposited on the island (Deposited) and the deposition rate.

Location	Method	Year	Period	Date	Eaten	Recovered	DE	Deposited	Deposition Rate (95% c.i.)
Miller Rocks	Volitional	2013	Nest building	4/12	103	12	0.75	16	15% (8-24%)
Miller Rocks	Volitional	2013	Incubation	5/10	99	5	0.80	6	6% (1-12%)
Miller Rocks	Volitional	2013	Chick	6/23	100	33	0.86	38	38% (27-50%)
Miller Rocks	Volitional	2013	AM	-	151	21	-	26	17% (10-24%)
Miller Rocks	Volitional	2013	PM	-	151	29	-	35	23% (16-31%)
Miller Rocks	Volitional	2013	Annual	-	302	50	-	61	20% (15-25%)
Blalock Is.	Volitional	2013	Nest building	4/18	101	8	0.76	11	10% (4-18%)
Blalock Is.	Volitional	2013	Incubation	5/11	101	7	0.80	9	9% (3-15%)
Blalock Is.	Volitional	2013	Chick	6/21	100	23	0.85	27	27% (17-37%)
Blalock Is.	Volitional	2013	AM	-	150	15	-	18	12% (7-18%)
Blalock Is.	Volitional	2013	PM	-	152	23	-	28	19% (12-26%)
Blalock Is.	Volitional	2013	Annual	-	302	38	-	46	15% (11-20%)
Crescent Is.	Volitional	2013	Nest building	4/14	99	9	0.59	15	15% (7-27%)
Crescent Is.	Volitional	2013	Incubation	5/6	101	5	0.67	7	7% (2-15%)
Crescent Is.	Volitional	2013	Chick	6/14	100	16	0.79	20	20% (11-30%)
Crescent Is.	Volitional	2013	AM	-	152	16	-	24	16% (9-24%)
Crescent Is.	Volitional	2013	PM	-	148	14	-	19	13% (7-19%)
Crescent Is.	Volitional	2013	Annual	-	300	30	-	43	14% (10-20%)
Island 20	Volitional	2013	Nest building	4/13	99	16	0.68	23	24% (14-36%)
Island 20	Volitional	2013	Incubation	5/5	98	5	0.74	7	7% (1-13%)
Island 20	Volitional	2013	Chick	6/13	100	12	0.83	14	14% (7-23%)
Island 20	Volitional	2013	AM	-	148	18	-	25	17% (10-24%)

Island 20	Volitional	2013	PM	-	149	15	-	20	13% (8-21%)
Island 20	Volitional	2013	Annual	-	297	33	-	45	15% (10-20%)
Combined	Volitional	2013	Annual	-	1,201	151	-	195	16% (14-19%)

Appendix B: Previously Reported Caspian Tern Predation Rates Adjusted for PIT Tag Deposition Rate, 2007-2013

This appendix provides historical predation rate estimates, those corrected for detection efficiency and deposition, for all Caspian tern colonies previously analyzed by Bird Research Northwest during 2007-2013 (Table B1). Methods described in Section 1.4 were used to calculate predation rates. All estimates use an on-colony PIT tag deposition rate of 0.71 (95% c.i. = 0.62 - 0.81; see BRNW 2013 for additional details) and colony-specific detection efficiencies (Table 3 or BRNW Annual Reports for 2008-2013).

Table B1. Estimated predation rates by Caspian terns on various Columbia Basin salmonid distinct population segments (DPSs) or evolutionarily significant units (ESUs) during 2007-2013, adjusted for on-colony PIT tag detection efficiencies and PIT tag deposition rates. Dashes indicate years when sample sizes of interrogated smolts at an upstream dam were too small (< 500 interrogated fish) for analysis. NA denotes colonies that were not scanned for PIT tags in that year. Values in parentheses are 95% confidence intervals.

Salmonid ESU[1]	Status[2]	2007	2008	2009	2010	2011	2012	2013
Predation by Caspian Terns on East Sand Island in the Columbia River Estuary are based on PIT-tagged smolts last interrogated at Bonneville or Sullivan dams								
SR Chinook $_{fall}$	T	3.3% (2.3-4.3)	1.9% (1.6-2.1)	2.0% (1.7-2.3)	0.7% (0.6-0.9)	0.7% (0.5-0.9)	0.7% (0.5-0.9)	0.8% (0.5-1.3)
SR Chinook $_{s/s}$	T	3.1% (2.8-3.5)	2.5% (2.1-2.9)	4.6% (4.2-5.2)	3.4% (3.0-3.8)	2.4% (1.9-3.0)	2.2% (1.8-2.7)	1.1% (0.8-1.5)
SR Sockeye	E	-	-	1.2% (0.6-1.9)	1.5% (0.7-2.4)	0.2% (<0.1-0.7)	2.1% (1.1-3.2)	0.7% (0.2-1.5)
SR Steelhead	T	22.6% (20.6-24.8)	14.3% (13.2-15.6)	14.6% (13.5-16.0)	14.0% (12.8-15.4)	11.8% (10.4-13.6)	10.0% (8.4-11.9)	12.5% (10.4-15.1)
UCR Steelhead	T	15.6% (13.7-17.7)	16.5% (14.5-18.8)	19.5% (17.1-22.3)	13.8% (12.4-15.2)	9.1% (7.3-11.1)	7.4% (6.0-9.1)	8.6% (7.1-10.6)
UCR Chinook $_{spr}$	E	1.9% (1.2-2.6)	1.7% (0.9-2.5)	3.6% (2.6-4.8)	2.9% (2.3-3.5)	2.6% (1.2-4.4)	1.2% (0.7-1.7)	0.6% (0.2-1.2)
UCR Chinook $_{s/f}$	NW	2.1% (1.3-3.1)	2.7% (1.9-3.5)	2.7% (1.9-3.5)	2.0% (1.7-2.3)	1.1% (0.7-1.6)	1.4% (0.9-2.0)	1.4% (0.8-2.0)
LW Sockeye	NW	2.1% (0.9-3.5)	0.8% (<0.1-1.7)	1.0% (0.2-1.9)	-	-	-	-
MCR Chinook $_{spr}$	NW	1.7% (1.2-2.3)	4.2% (3.4-5.0)	3.5% (2.8-4.3)	4.6% (3.9-5.3)	1.9% (1.3-2.7)	1.6% (1.0-2.2)	1.3% (0.7-1.9)
MCR Steelhead	T	18.4% (16.1-21.0)	13.6% (11.6-15.8)	14.1% (12.2-16.2)	11.9% (10.6-13.3)	9.6% (6.8-12.8)	9.3% (6.7-12.3)	9.6% (7.1-12.5)
UWR Chinook $_{spr}$	T	1.3% (0.7-2.1)	4.4% (3.4-5.5)	1.7% (1.3-2.2)	1.5% (0.3-3.2)	0.8% (0.2-1.5)	0.7% (0.4-1.1)	0.9% (0.4-1.4)
UWR Steelhead	T	-	-	-	-	-	-	-
DR Chinook $_{s/f}$	NW	-	-	-	-	-	-	-
OR Sockeye	NW	-	-	-	-	-	-	-
Predation by Caspian Terns on the Blalock Islands in the Columbia River are based on PIT-tagged smolts last interrogated at McNary Dam								
SR Chinook $_{fall}$	T	0.1% (<0.1-0.2)	0.1% (0.1-0.1)	<0.1%	<0.1%	0.1% (0.1-0.2)	NA	<0.1%
SR Chinook $_{s/s}$	T	0.1% (<0.1-0.1)	0.1% (0.1-0.2)	0.3% (0.2-0.3)	0.1% (<0.1-0.1)	0.1% (<0.1-0.1)	NA	<0.1%)
SR Steelhead	T	0.9% (0.6-1.1)	0.7% (0.6-0.9)	0.6% (0.5-0.7)	0.9% (0.7-1.1)	0.1% (<0.1-0.2)	NA	0.1% (<0.1-0.2)
UCR Steelhead	T	0.9% (0.5-1.4)	0.6% (0.3-1.0)	0.5% (0.2-0.8)	0.9% (0.5-1.3)	0.1% (<0.1-0.2)	NA	<0.1%
UCR Chinook $_{spr}$	E	<0.1%	<0.1%	0.2% (<0.1-0.3)	<0.1%	<0.1%	NA	<0.1%

UCR Chinook $_{s/f}$	NW	0.1% (<0.1-0.2)	0.3% (0.1-0.4)	<0.1%	0.1% (0.1-0.2)	<0.1%	NA	<0.1%
SR Sockeye	E	-	-	<0.1%	0.1% (<0.1-0.4)	0.3% (0.1-0.6)	NA	<0.1%)
LW Sockeye	NW	<0.1%	<0.1%	<0.1%	<0.1%	-	NA	-
OR Sockeye	NW	-	-	-	-	-	NA	-
Predation by Caspian Terns on Crescent Island in the Columbia Rivers are based on PIT-tagged smolts last interrogated at Lower Monumental or Rock Island dams								
SR Chinook $_{fall}$	T	0.8% (0.4-1.4)	1.5% (1.3-1.8)	1.0% (0.9-1.2)	0.9% (0.8-1.1)	0.5% (0.4-0.6)	0.5% (0.4-0.7)	0.6% (0.4-0.9)
SR Chinook $_{s/s}$	T	0.4% (0.3-0.5)	0.9% (0.7-1.1)	1.4% (1.2-1.7)	0.4% (0.2-0.6)	0.7% (0.6-0.8)	0.6% (0.4-0.8)	0.5% (0.4-0.7)
SR Steelhead	T	3.9% (3.4-4.4)	5.8% (5.2-6.6)	4.5% (4.0-5.0)	3.9% (3.3-4.6)	2.6% (2.3-3.0)	2.8% (2.4-3.5)	2.8% (2.4-3.4)
UCR Steelhead	T	2.4% (1.8-3.1)	2.8% (2.2-3.4)	2.2% (1.7-2.7)	1.7% (1.3-2.1)	2.4% (1.9-2.9)	1.2% (0.8-1.6)	2.8% (2.2-3.5)
SR Sockeye	E	-	1.4% (0.3-2.7)	0.9% (0.5-1.4)	1.2% (0.3-2.6)	0.7% (0.5-0.9)	1.3% (0.9-1.8)	0.5% (0.1-1.0)
UCR Chinook $_{s/f}$	NW	-	-	0.1% (<0.1-0.4)	0.1% (<0.1-0.2)	0.2% (<0.1-0.4)	<0.1%	<0.1%
UCR Chinook $_{spr}$	E	-	-	<0.1%	0.6% (<0.1-1.5)	0.4% (<0.1-0.9)	0.1% (<0.1-0.4)	0.2% (<0.1-0.6)
LW Sockeye	NW	-	-	-	-	-	-	-
OR Sockeye	NW	-	-	-	-	-	-	-
Predation by Caspian Terns on Goose Island in Potholes Reservoir, WA, are based on PIT-tagged smolts last interrogated Lower Monumental or Rock Island dams								
SR Chinook $_{fall}$	T	0.2% (<0.1-0.8)	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
SR Chinook $_{s/s}$	T	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
SR Steelhead	T	<0.1%	<0.1%	0.1% (<0.1-0.1)	<0.1%	<0.1%	0.2% (0.1-0.3)	0.1% (<0.1-0.2)
UCR Steelhead	T	12.9% (8.5-20.6)	10.7% (9.3-12.2)	22.0% (19.1-25.9)	13.7% (11.8-16.3)	12.6% (10.6-15.1)	17.3% (14.1-21.7)	14.9% (12.7-17.8)
SR Sockeye	E	-	0.3% (<0.1-0.8)	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.2)	<0.1%
UCR Chinook $_{s/f}$	NW	-	-	0.3% (<0.1-0.9)	0.3% (<0.1-0.6)	0.1% (<0.1-0.3)	0.1% (<0.1-0.2)	0.3% (0.1-0.6)
UCR Chinook $_{spr}$	E	-	-	5.0% (2.1-8.6)	1.4% (0.3-2.9)	0.5% (<0.1-1.3)	2.5% (1.0-4.4)	2.1% (0.7-4.0)
LW Sockeye	NW	-	-	-	-	-	-	-
OR Sockeye	NW	-	-	-	-	-	-	-

[1] DR = Deschutes River, LW = Lake Wenatchee, MCR = Middle Columbia River, OR = Okanagan River, SR = Snake River, UCR = Upper Columbia River, UWR = Upper Willamette River

[2] E = Endangered, T = Threatened, NW = Not Warranted

APPENDIX C: Foraging Behavior and Dispersal Patterns of Caspian Terns Nesting on Goose Island, Potholes Reservoir, Washington

This study was designed to evaluate the foraging behavior of Caspian terns (*Hydroprogne caspia*) nesting on Goose Island in Potholes Reservoir, with particular emphasis on predation by terns from this colony on ESA-listed juvenile salmonids (*Oncorhynchus* spp.) from the upper Columbia River. The Goose Island tern colony is one of the largest Caspian tern colony in the Columbia Plateau region; colony size was estimated at ca. 460 and ca. 340 breeding pairs in 2012 and 2013, respectively. Goose Island terns commute to the middle Columbia River to consume juvenile salmonids and losses of ESA-listed Upper Columbia steelhead (*O. mykiss*) to predation by Caspian terns nesting at Goose Island have been substantial; predation rates in excess of 10% of the available steelhead population below Rock Island Dam have been recorded (Evans et al. 2012). We assessed foraging habitats used by Caspian terns nesting at Goose Island using remotely downloadable GPS tags.

Avian predation is a factor limiting the recovery of some salmonid populations from the Columbia River basin that are listed under the Endangered Species Act (ESA; Collis et al. 2002; Lyons et al. 2011; Evans et al. 2012). Caspian terns, double-crested cormorants (*Phalacrocorax auritus*), American white pelicans (*Pelecanus erythrorhynchos*), and several species of gulls (*Larus* spp.) have all been identified as predators of anadromous juvenile salmonids in the Columbia Plateau region. Of these avian predators, Caspian terns have been reported as having the highest per capita (per bird) impacts on survival of juvenile salmonids, especially steelhead, a salmonid species shown to be particularly susceptible to tern predation (Collis et al. 2001; Antolos et al. 2005; Evans et al. 2012).

Caspian terns are colonial fish-eating waterbirds that nest along coastlines, in estuaries, and at inland sites along large rivers and lakes (Cuthbert et al. 1999). The breeding season for Caspian terns is generally from April to July (Cuthbert et al. 1999). Caspian terns are considered strictly piscivorous and forage by plunge-diving into the water to capture fish near the surface. Records of Caspian terns nesting in southeastern Washington in the Columbia Plateau region date back to 1929 (Kitchin 1930), when a small colony of Caspian terns was observed nesting on Moses Lake, WA. Recently, Adkins et al. (2011) reported five Caspian tern colonies in the Columbia Plateau region during 2004-2009, ranging in size from an average of six pairs on Sprague Lake, WA to an average of 424 pairs on Crescent Island in the McNary Dam Reservoir.

One of the largest Caspian tern colonies in the Columbia Plateau region is located on Goose Island in Potholes Reservoir, WA, with an average colony size of ca. 400 breeding pairs during 2008-2011 (BRNW 2012). Caspian terns nesting on Goose Island commute > 30 km from Potholes Reservoir to the middle Columbia River to consume anadromous juvenile salmonids (Maranto et al. 2010). Studies of the diet composition of Caspian terns nesting at Goose Island indicate that salmonids comprise a minority of prey items in the diet, with roughly 10% to 30% of the fish consumed each year consisting of juvenile salmonids (Maranto et al. 2010; BRNW 2012). Despite the low overall percentage of salmonids in the diet of terns nesting at Goose

Island, losses of ESA-listed juvenile steelhead to Goose Island Caspian terns have been substantial. Evans et al. (2012) estimated that minimum predation rates (number consumed/number available) on ESA-listed steelhead from the Upper Columbia River population were in excess of 10% of the population available below Rock Island Dam during 2008 - 2010. Lyons et al. (2011) estimated that the annual population growth rate (λ) of Upper Columbia River steelhead could be increased by 4.2% (for the hatchery-raised portion of the population) and 3.2% (for the wild portion of the population) if predation by Caspian terns nesting at the Goose Island colony was eliminated.

Survival standards for juvenile salmonids established under the 2004 Biological Opinion for the Priest Rapids Project (Wanupum Dam and Priest Rapids Dam and their associated reservoirs) require at least 93% survival for juvenile salmonids through each hydropower development (one dam and reservoir; NMFS 2004). To evaluate whether these standards were being met, Grant County Public Utility District (GPUD) No. 2 conducted salmonid survival studies from 2008 to 2010 within the Priest Rapids Project. Survival studies utilized double-tagged (acoustic tag and passive integrated transponder [PIT] tag) steelhead smolts to track fish behavior (travel times and routes) and estimate survival (Timko et al. 2011). Results indicated that survival standards for steelhead were not being met in the Priest Rapids development during 2008-2010 and in the Wanapum development in 2010 (Thompson et al. 2012). Estimates of predation rates by Goose Island Caspian terns on steelhead smolts tagged and released by GPUD during these years ranged from 12.8% to 20.8% of available steelhead smolts below Rock Island Dam (Evans et al. 2013), indicating that predation by Caspian terns was a substantial source of smolt mortality in the Project.

Resource management agencies are currently proposing a management plan aimed at reducing the impacts of Caspian terns that currently nest in the Columbia Plateau region (i.e., colonies on Goose Island and Crescent Island) on the survival of ESA-listed salmonids, in particular, steelhead smolts from the Upper Columbia River and Snake River populations. Management initiatives under consideration are focused on reducing tern predation on Columbia Basin salmonids without adversely affecting the Caspian tern population in western North America, which may require (1) redistribution of Caspian terns from breeding colony sites in the Columbia Plateau region to multiple dispersed colony sites elsewhere within their breeding range (USFWS 2005) and (2) identifying specific sites on the mid-Columbia River where tern predation activity is high and implementing measures to protect smolts in those locales. Currently it is unknown to what extent terns nesting on Goose Island in Potholes Reservoir are utilizing the middle Columbia River for foraging and where along the middle Columbia River the impacts from tern predation are highest.

Advances in GPS-telemetry have created new opportunities to study individual behavior and movements, as well as how animals interact with features of their environment (Schick et al. 2009, Cagnacci et al. 2010). We used micro-GPS transmitters weighing less than 15 g with remote download capabilities to collect fine-scale movement data on breeding Caspian terns over multiple days without having to retrieve the data logger. This allowed us to continuously track breeding Caspian terns and provide a complete profile of individual foraging trips and daily movements. By using cluster analysis to infer behavioral state from movement data (Van Moorter et al. 2010, Patterson 2012) we can identify areas where terns are concentrating search activity, quantify activity rates for whole trips and days, and examine how terns commute between the colony and foraging areas.

Our objectives in this study were to understand how terns nesting at Goose Island use the middle Columbia River for foraging during the breeding season. We wanted to describe foraging and commuting strategies of Caspian terns nesting at Goose Island. It is currently unknown what portion of Caspian terns nesting on Goose Island commute to the Columbia River to forage or how these long-distance foraging trips differ from foraging closer to the colony. Our second objective was to identify any foraging hot spots on the middle Columbia River where juvenile salmonids are especially susceptible to Caspian terns and other avian predators. While it is well-documented that Goose Island Caspian terns consume Columbia River anadromous juvenile salmonids, the specific locations where this occurs are largely unknown. The spatial resolution available with GPS tag technology could provide detailed information on habitat types and specific locations where smolts are particularly susceptible to predation by Caspian terns and other fish-eating birds. Finally, we wanted to determine if terns foraged in water bodies between Potholes Reservoir and the Columbia River while commuting to and from the Columbia River. Diet composition of Goose Island Caspian terns is quantified by observing the fish carried back to the colony by adult terns to feed their mates and chicks. If terns frequently catch a non-salmonid prey item on their way back from the Columbia River and feed that to their mate or chick, this would cause an under-estimation of the proportion of juvenile salmonids in the tern diet and bioenergetics-based estimates of smolt consumption would be biased low.

METHODS

GPS Tagging

On 20 May we deployed GPS transmitters on 28 adult Caspian terns: 19 males and 9 females. We used 24 Telemetry Solutions transmitters (44 x 22 x 16 cm, 14 g; Concord, CA) and 4 Skorpa Telemetry transmitters (44 x 22 x 15 cm, 17 g; Aberfeldy, Scotland). GPS units were attached to the base of the four central rectrices (tail feathers) with glue and cable ties. Transmitters and attachments weighed an average of 2.2% of tern body mass (maximum = 2.7%).

We collected 5-7 breast feathers from each bird for DNA-sexing; analysis was conducted by Avian Biotech International (Tallahassee, FL). Terns outfitted with GPS transmitters were given a pink mark on the breast and back so researchers could easily identify them on colony. Locations of nests belonging to tagged terns were identified during the first day of tracking and breeding status was monitored until nest completion. Breeding status for each individual was classified as incubating eggs, attending chicks, or failed, depending on the status of its nest at the end of each day.

GPS transmitters were programmed to record positions at 4-min intervals from 05:00 to 21:00 PDT. All transmitters were programmed to begin collecting data at 05:00 on 21 May. Some transmitters were turned off on 23 May and 24 May in order to spread data collection over a longer period. Data were downloaded remotely when terns returned to the colony. GPS transmitters would fall off when terns enter the post-season molt and molt their rectrices.

Behavioral Classification

GPS data were filtered to remove missed locations and locations that would require flight velocities greater than 80 km/hr. We chose 80 km/hr as a threshold for excluding points based on visual examination of a histogram of all velocities, and mapping locations with velocities greater than 70 km/hr. There were few locations with flight velocities greater than 80 km/hr. When mapped these locations appeared to represent an anomalous change in direction or velocity relative to the previous and subsequent locations.

Locations were considered active if there were three or more consecutive locations where the tern moved more than 100 m and the individual was at least 500 m from the colony. Velocity and turning angle at each active location were used as measures of movement characteristics (Calenge et al. 2009). We calculated velocity as the distance between the current location and the next location, divided by the time between locations. Turning angle was calculated as the change in direction between the previous and subsequent locations. Values of velocity ranged from 0 to 80 km/hr, and values of turning angle ranged from 0^o to 180^o.

For all active locations we used k-means cluster analysis to identify patterns of movement that represent distinguishable behavioral states (Van Moorter et al. 2010). Cluster analysis uses multivariate data (e.g., velocity and turning angle) to identify clusters of observations with similar characteristics (Steinley 2006). Analysis was performed for all possible numbers of clusters between 1 and 10, and we used the gap statistic (Tibshirani et al. 2001) to identify the optimum number of clusters in the data set (Van Moorter et al. 2010). The gap statistic estimates the number of groups within a data set by comparing the change in within-cluster dispersion for each number of clusters to the dispersion expected from simulated reference null distributions (Tibshirani et al. 2001). We performed range standardization on velocity and turning angle before analysis so that differences in range between variables would not affect the contribution of each variable to the clustering (Steinley 2006). The gap statistic was calculated from 50 simulated data sets and the tolerance level was set to 2; higher tolerance values increase the evidence necessary to include additional clusters (Van Moorter et al. 2010).

Foraging trips were defined as any set of five or more consecutive locations at least 500 m from the colony. Within trips, foraging bouts were defined as three or more consecutive foraging locations. For each trip we identified the primary destination (the furthest water body visited during a trip). We looked at the proportion of terns using different foraging areas, how many foraging destinations individual terns used, and the number of trips made per day in relation to primary trip destination. All areas within 20 km of the colony were treated collectively as the Potholes Reservoir area. We excluded four identified trips with a maximum distance from colony of less than 2.5 km, because video footage of the colony during two of these trips showed that these trips did not occur. Trips were considered complete if all gaps of more than three locations (16 min) did not add up to more than 25% of total trip duration.

To determine if foraging behavior differed between trips to different areas we compared foraging trip characteristics among trips based on primary destination. For each trip we calculated maximum distance from the colony (km), total trip duration (min), the number of foraging bouts, and time spent in different behaviors (foraging, commuting, and resting). We used mixed-effects models to compare trip duration and time spent in different behaviors as a function of primary destination. Tern identity was included as a random effect to account for lack of independence among trips made by the same individual. A variance co-efficient was included to allow variance to change with maximum distance from colony. The number of foraging bouts per trip is count data; therefore, we used a generalized linear mixed model (GLMM) with a poisson distribution in the analysis of the number of foraging bouts. Plots of standardized residuals against fitted values and normal quantile-quantile plots were used to confirm that these models met the assumptions of normality and homogeneity of variance (Bolker *et al.* 2008, Zuur *et al.* 2009). The approximate estimate of over-dispersion in the poisson GLMM was <1, indicating that over-dispersion was likely not an issue in this model.

Preliminary examination of the data indicated that there could be sex-specific differences in foraging strategy. Trips were classified as either short (< 30 km from the colony) or long (> 30 km from the colony). A binomial GLMM was used to test whether or not females were more likely to make long trips than males. Tern identity was included as a random effect to account for lack of independence in trips made by the same individual. We assessed model fit using a simulated logistic regression quantile-quantile plot following the method described in Zuur *et al.*(2009). Briefly, 1000 data sets were simulated from the fitted model to develop an expected distribution of residuals assuming this model is true. The observed residuals were compared to the simulated distribution to determine if there were any significant deviations from the model assumptions.

We used Spearman's correlation tests to examine the relationships between proportions of time spent in four different behaviors each day: on colony, foraging, commuting, and resting off colony. We calculated average time spent in these four behavioral categories for each tern, based on all complete days of data collected for that individual.

Foraging at Secondary Sites

If there were multiple foraging bouts in a trip that used different water bodies, this was considered the secondary destination. We examined all trips outside of the Potholes Reservoir area to determine the proportion and characteristics of trips that included a secondary foraging area.

Foraging bouts represent a sustained period of concentrated search. If terns are opportunistically foraging for prey to bring back to the nest as they return from the nest, this search may be too short in duration to be picked up as a foraging bout, given the temporal resolution of our data. We calculated a straightness index for the outgoing and incoming portions of each trip to determine if smaller scale foraging occurs along the way to or from primary foraging areas. The outgoing and incoming portions of a trip were defined as all locations within 75% of the maximum distance from colony at the start (outgoing segment) and end (incoming segment) of each trip. Straightness was calculated as the ratio of the total distance travelled between GPS locations (the tern's path) and the straight line distance between the colony and the furthest point in that commuting segment. This index can take values from 1 to 0: a value of 1 would be a straight line and values closer to 0 represent more sinuous paths. We would expect terns foraging during the commute to have a more sinuous path and a lower straightness value. We used a generalized linear mixed-effects model with a binomial distribution and random effect of individual to test whether there was a difference in straightness between the out-going and incoming trip segments. Analysis was limited to complete trips and trip segments with at least three locations.

Core Use Areas

We estimated three types of core use areas (foraging, resting, and commuting) using the kernel density tool in ArcGIS (ArcMap v. 10.0). Estimates for core foraging and resting areas were each based on all recorded locations classified as foraging (n = 1337) or resting (n = 1771) that were at least 2.5 km from the colony; we expected resting and foraging to occur adjacent to the colony, but wanted to focus on identifying sites away from the colony. Core commuting areas were based on all commuting locations that were at least 15 km from the colony. We were primarily interested in identifying the routes terns used to reach more distant foraging areas. The search radii for each estimate was based on the north-south extent of the point file divided by 30 (3877 m for foraging, 3221 m for resting, and 3763 m for commuting), and output cell size was 20 m.

Nest Success

During the GPS tag deployment nests of tagged terns were monitored twice daily. After all GPS tags failed, nests were monitored during each visit to the colony. In addition to the nests of GPS tagged terns, we monitored 25 nests of terns that were captured and banded but not tagged. These control nests were used to determine whether GPS tagging had a negative impact of nest success. During each monitoring session we recorded the presence of an adult tern attending the nest and the contents of the nest, if possible. It is not always possible to monitor nests once chicks become mobile; therefore, we considered nests successful if chicks reached 21 days post-hatch. We used Fisher's exact tests to determine if there was a difference in the hatch rate or nest success rate between tagged and control nests or among terns that used different foraging areas.

Bill Load Identification

During the GPS deployment one observer was present in an observation blind adjacent to the colony at all times to identify GPS-tagged terns returning to the colony with fish. When a tagged tern returned with a bill load to the colony, the observer would identify the fish to species and estimate fish length. Observers would continue to follow the individual tern until the fish was consumed and the tern identity had been determined by the alphanumeric band or nest location. Two cameras were placed on the colony prior to the breeding season. Following capture the locations of any nests belonging to GPS-tagged terns in front of the cameras were mapped. We reviewed video footage of these nests for up to four days following GPS deployment (or as long as the GPS tag was functioning) to identify any fish brought back to the nest. Any bill load fish identified in real time or on video footage were cross-referenced with GPS tracking data to determine where terns could have foraged for that species.

Statistical Analysis and Mapping

Statistical analyses were conducted in R version 2.13.2 (<http://www.r-project.org/>). Mixed-effects models were conducted using 'nlme' (Pinheiro et al. 2011) and GLMMs were conducted using 'lme4' (Bates et al. 2011). Kernel density estimates and maps were conducted in ArcGIS version 10.0.

RESULTS

GPS Data Collection

We were able to retrieve data from 23 of the GPS transmitters deployed on Caspian terns nesting at Goose Island in Potholes Reservoir (Figure C1); tracking periods for each transmitter ranged from less than 6 hours to 5 days. We collected 53 complete days of data from 20 terns, between 21 May and 27 May (Table C1). We also retrieved 14 partial days of location data from tags that stopped working during the middle of the day. Due to tag failure we were only able to retrieve a partial day of data for three individuals and we were unable to retrieve any data from five transmitters.

GPS data were collected for 9 females and 13 males (Table C2). Most terns were attending nests with eggs throughout the tracking period. Three individuals had chicks for all or part of the time tracked. One GPS-tagged individual was not actively nesting (Table C2). We restricted all statistical analysis to terns attending nests with eggs because of the small number of individuals tracked with chicks or after nest failure.

Behavioral Classification

Cluster analysis identified three movement states as the optimal clustering of the behavioral data. The three movement states differed in both velocity (*F* = 825 df = 2 and 2484, p-value < 0.001) and turning angle (*F* = 7671, df = 2 and 2484, p-value < 0.001). Based on the characteristics of the three movement states, we classified them as either (1) commuting, (2) extensive search, and (3) intensive search (Figure C2 and Appendix D, Figure D3). Commuting movements were characterized by high velocities (median = 46 km hr^{-1}, range = 30 – 80 km hr^{-1}) and low turning angles (median = 8^o, range = 0 – 103^o); these fast, directed movements occurred when a tern was commuting between areas. Extensive search movements were characterized by low velocities (median = 18 km hr^{-1}, range = 0 – 40 km hr^{-1}) and moderate turning angles (median = 27^o, range = 0 – 93^o); these slow, directed movements occurred when a tern was searching an area slowly, without doubling back on itself. Intensive search movements were characterized by even lower velocities (median = 10 km hr^{-1}, range = 0 – 73 km hr^{-1}) and high turning angles (median = 150^o, range = 88 – 180^o); these tortuous movements occurred when a tern was making slow, tight turns over a small area. For all subsequent analysis, intensive search and extensive search were considered collectively as foraging behavior.

Foraging Strategies

We recorded 97 foraging trips by 22 terns (9 females and 13 males, Table C2). Half of the terns made at least one trip to the middle Columbia River and 32% of the terns made foraging trips to the Columbia River exclusively (Table C2). Just over half of the terns (55%) made foraging trips within the Potholes Reservoir area and 23% used the Potholes area exclusively (Table C2). Three terns (14%) made foraging trips to the Snake River (Table C2). Three individuals (14%) used foraging areas that were located between Potholes Reservoir and the two rivers (Eagle Lakes, Scootenay Reservoir, and Columbia National Wildlife Refuge). There was a high degree of consistency in primary destination: only 5 of 19 individuals with multiple foraging trips used two primary destinations (Table C2).

When terns foraged exclusively in the Columbia River, they generally made one trip to the river per day (84%) and never made more than two trips in a day (Table C3). Terns that foraged only in the Potholes Reservoir area made two or three trips per day (42% and 47%, respectively; Table C3). Terns that made foraging trips to the Snake River only made one trip per day (Table C3). There were three males attending eggs that did not make any foraging trips during the day for at least one day of data collection. The three terns attending nests with chicks all foraged within the Potholes Reservoir area and made 2 – 3 trips per day.

The majority of foraging trips recorded (60%) occurred within the area around Potholes Reservoir (within 20 km of the colony; Table C2). These trips took significantly less time than trips to the Columbia River or the Snake River (95% c.i. = 71 – 134 min) and included fewer foraging bouts (95% c.i. = 1.15 – 1.81; Table C4). Foraging trips within the Potholes Reservoir area occurred in the reservoir, Winchester Wasteway, Frenchman Hills Wasteway, Seep Lakes, and Hiawatha Lake. Trips within the Potholes area included less than half as much time commuting (95% c.i. = 16 – 31 min) as trips to the Columbia River or the Snake River. These trips to the Potholes area also included less foraging time (95% c.i. = 44 – 61 min) and resting time (95% c.i. = 4 – 59 min) than did trips to the Columbia or Snake rivers.

There were 32 trips to the middle Columbia River, ranging from Rock Island Dam to 10 km below Hanford Reach (Table C2). Foraging locations along the middle Columbia River were concentrated in Hanford Reach, Priest Rapids Reservoir, and Wanapum Reservoir (up to 25 km upstream of Wanapum Dam). On average, trips to the Columbia River took 4.5 to 6.5 hrs (95% c.i. = 277 – 391 min), and included multiple foraging bouts (95% c.i. = 1.87 – 3.13). Trips to the Columbia River included more time foraging (95% c.i. = 94 – 139 min) than trips within the Potholes Reservoir area. Times spent commuting (95% c.i. = 78 – 102 min) and resting (95% c.i. = 80 – 160 min) were longer compared to foraging trips in the Potholes Reservoir area, but shorter compared to foraging trips to the Snake River.

We recorded four trips to the Snake River by three different individuals: two terns attending nests with eggs and one that was not actively nesting (Table C2). One nesting tern was tracked for two days and made one trip to the Snake River on each day. The other nesting tern was tracked for three days; on the first day it went to the Snake River and on the other two days it remained on-colony all day. Trips to the Snake River had maximum distances from the colony of 87 – 93 km and took most of the day (95% c.i. = 550 – 912 min), longer than trips to any other area. Foraging on the Snake River occurred from Lyon's Ferry Hatchery to 36 km downstream of Lower Monumental Dam. Trips to the Snake River included more foraging bouts (95% c.i. = 3.75 – 7.45), more time commuting (95% c.i. = 115 – 191 min), and more time resting (95% c.i. = 285 – 524 min) than trips to other areas. The amount of time spent foraging during trips to the Snake River was similar to trips to the Columbia River (95% c.i. = 98 – 253 min).

Females were more likely than males to make long foraging trips ($\chi^2 = 6.94$, p-value = 0.008). There was a 96.3% (95% c.i. = 68.6 – 99.7%) chance that a trip by a female tern would be more than 30 km from the colony. Males had only a 25% (95% c.i. = 5.4 – 66.0 %) chance of making a trip more than 30 km from the colony.

On average, Caspian terns attending nests with eggs spent the majority of each day on colony (63%, Figure C3), and similar proportions of each day commuting (10%), foraging (12%), and resting off colony (15%). The proportion of time terns spent foraging was not related to any other behavior (Table C5). There was a positive relationship between time spent commuting and time spent resting off colony (Table C5). Time spent on colony decreased as time spent commuting and time spent resting off colony increased (Table C5).

Foraging at Secondary Sites

Out of 39 trips with a primary destination outside of the Potholes Reservoir area, only 18% (7 trips) included a foraging patch on the way to or from the primary destination. Six trips included a foraging patch on the way to the primary foraging sites in the Columbia River, Snake River, or Scootenay Reservoir. Only one trip included foraging patches during the return portion of a trip from the middle Columbia River. Foraging bouts on the way to or from the primary destinations were short components (16 – 36 min) of much longer trips (> 4.5 hrs.). Two of these trips had two foraging bouts away from the primary foraging destination; the other five trips had only one foraging bout along the route. Only one tern was recorded making this type of trip on multiple occasions.

There was no difference in path straightness between out-going and in-coming trip sections ($\chi^2 = 0.59$, p-value = 0.441). Average path straightness for the out-going and incoming trip sections was 0.96 (95% c.i. = 0.90 – 0.98). This indicates that the commuting portion of foraging trips included some search behavior, but on average the path taken by terns was only 4% longer than the most direct route possible.

Core Use Areas

We identified eight core foraging areas: three along the middle Columbia River, one on the lower Snake River, and four close to or within Potholes Reservoir (
Figure). The most individual terns foraged in Wanapum Reservoir just north of Wanapum Dam. Priest Rapids Reservoir and the Hanford Reach were the two other core foraging areas along the middle Columbia River. One core foraging area, used by two terns, was identified on the lower Snake River: about 15 km upstream from Lower Monumental Dam and 8 km downstream from the Lyons Ferry Fish Hatchery. Close to the colony most terns foraged in the Winchester Wasteway and the northeast portion of Potholes Reservoir. Core foraging areas were also identified at Seep Lakes and Frenchman Hills.

There were seven core resting areas: four along the middle Columbia River, one on the Snake River, and two close to Potholes Reservoir (
Figure). In the Columbia River the three most used resting areas were in Priest Rapids Reservoir (Cabin Island and a sand bar just west of Desert Aire) and the Hanford Reach (sandbar upstream of Locke Island). There was also an important resting area in Wanapum Reservoir on an island just south of the Vantage Bridge. All resting sites identified overlapped with core foraging areas. The resting areas used by the most terns were on the middle Columbia River and in the Winchester Wasteway.

The most important commuting route between the colony and middle Columbia River, used by eight terns, ran east-west between Potholes Reservoir and the Wanapum Dam (
Figure). This route passed over the Frenchman Hills Wasteway, but did not follow Lower Crab Creek. There was a commuting route running north-south from Potholes Reservoir to the

Hanford Reach and a commuting route connecting Wanapum Dam and Priest Rapids Dam. There was also an important foraging route running from Potholes Reservoir to the Winchester Wasteway.

Nesting Success

Terns tagged with GPS transmitters had a similar hatching rate (68%) and fledging rate (29%) as control terns that were captured, but not tagged (72% hatched and 24% fledged; hatch: p-value = 1.00, fledge: p-value = 1.00). Nests of terns that foraged exclusively in the Potholes Reservoir area hatched (88%) and fledged (38%) at a slightly higher rate than did terns that made some trips to the Columbia River (64% hatched and 18% fledged) or the Snake River (33% hatched and 0% fledged). These apparent differences were not statistically significant (hatch: p-value = 0.80; fledge: p-value = 0.81).

Bill Load Identification

During the GPS deployment we identified 33 bill load fish brought back to the colony by GPS-tagged terns; unfortunately, only five of these identified bill loads were for terns with an active GPS tag. These bill loads were sunfish (2), bass (1), and yellow perch (2); none of these foraging trips included travel to the Columbia or Snake rivers. We reviewed 22 days of video footage from seven nests with a tagged adult. Only one bill load was brought back to the nest (by an untagged tern), indicating that there was very limited mate feeding activity taking place during the tagging period.

DISCUSSION

Over half (64%) or the terns tracked from the Goose Island colony did some foraging in either the middle Columbia River or lower Snake River and nearly half (45%) only foraged at one of the two rivers during the time they were tracked. Most terns consistently foraged either at one of the rivers or in the area around Potholes Reservoir during the period of tracking. Only 21% of terns tracked for multiple trips had trips to both the Columbia River and trips within the Potholes Reservoir area. Foraging trip characteristics varied depending on trip destination; trips to the Columbia and Snake rivers were longer in duration, included proportionately more time commuting, foraging, and resting, and had more foraging bouts per trip. This indicates terns compensate for the additional time required to commute to distant foraging areas by spending more time foraging at those locations. Terns foraging at the rivers made fewer trips per day, which also supports the idea that terns foraged more during these trips to offset the extra time required.

Females were more likely to make a long-distance foraging trip (> 30 km from the colony) compared to males while attending a nest with eggs. Studies of foraging distribution of Caspian terns at other colonies have not found any difference in foraging distribution between males and females (Anderson et al. 2007; Adrean 2011; Patterson 2012). There is some evidence that foraging behavior (number of trips per day and time spent on colony) can be different for males and females, depending on prey availability (Adrean 2011). Females may have higher energetic requirements than males following egg-laying and could benefit from abundant, high-quality prey available at the Columbia River. In this case we would expect females to forage primarily at the rivers during the incubation period in all years.

We only tracked three individuals attending nests with chicks, so it is not possible to draw any conclusions about foraging behavior during this portion of the breeding cycle. Anecdotally, all three terns with chicks (1 female and 2 males) foraged exclusively within the Potholes Reservoir area, indicating that nesting terns may alter their foraging behavior after hatching. We would expect terns to forage closer to the colony while attending small chicks (Anderson et al. 2007); energy requirements increase with the need to provision chicks, and foraging closer to the colony allows adults to provision chicks more frequently. Foraging behavior probably varies throughout the season in response to changing energy demands and changes in the distribution and availability of prey.

Two of the four core foraging areas on the Columbia and Snake rivers were located in the reservoirs behind Wanapum and Priest Rapids dams. Reservoirs behind Lower Monumental, Ice Harbor, and Rock Island dams were not identified as core foraging areas, even though these sites were within the foraging range of Goose Island terns. All of the core foraging areas on the rivers were associated with important resting areas. There is no way to determine whether terns choose resting areas because of their proximity to foraging areas or vice versa. The core foraging area on the Snake River was located around some of the only islands between the Lyons Ferry hatchery and Lower Monumental Dam, suggesting this area may be used because terns can forage close to a resting site. In the Hanford Reach, foraging and resting areas are located on islands at a major bend in the river, even though there are other islands both upstream and downstream. River flows through this bend could make it easier to catch fish. This is also the closest section of the Columbia River to the Goose Island colony; terns may forage and rest here because it is convenient.

Small numbers of PIT tags from Snake River salmonids have been found on Goose Island in previous years (BRNW 2012). It had been assumed that these tags were deposited by failed breeders or non-breeding migrants because the Snake River is so far from the colony. GPS tracking data show that a small percentage of the terns breeding at Goose Island are foraging on the Snake River, at least during the late incubation stage of nesting. Foraging trips to the Snake River extended to 87 – 93 km straight-line distance from the colony, the longest ever recorded for an actively nesting Caspian tern (Adrean 2011, Patterson 2012). These long-distance foraging trips have implications for management planning, if alternative nesting habitat for Caspian terns is within this extended foraging range of critical salmon habitat. Trips to the Snake River were longer in duration than trips to any other foraging area; in each instance terns making these trips were away from the colony for most of the day. This would restrict foraging opportunities for the mate and limit the ability of terns with chicks to provision chicks regularly. For this reason, it is likely that terns making trips to the Snake River have to change foraging strategies once chicks hatch.

Long trips to the Snake River might be advantageous during the incubation period if terns are able to capture prey more efficiently and/or capture prey with a higher energy content than at other foraging areas. If inter- or intra-specific competition for forage fish is high closer to the colony, some terns may be forced to forage further from the colony, either to increase capture efficiency or to avoid piracy and kleptoparasitism. In this scenario, we would expect to see more foraging at the Snake River (and more Snake River PIT tags) in years when prey availability is low or the numbers of terns or gulls nesting at Goose Island are high. Some individuals may continue to commute to distant foraging areas that they have used in the past (i.e., during the pre-breeding season, while breeding at another site, or during previous seasons as a non-breeder or failed breeder). High foraging site fidelity that persists across seasons could affect management efforts if nesting habitat enhancement sites are within ~100 km of foraging areas currently used on the Columbia or Snake rivers, or if terns are reluctant to re-locate to nesting areas out of range from their known foraging areas. Alternatively, the terns we recorded making these long trips could be less experienced breeders that have not learned to balance self-maintenance with breeding effort.

There has been a discrepancy in previous years between the estimated number of salmonids consumed by terns nesting at the Goose Island colony based on identification of fish in bill loads on-colony and recoveries of smolt PIT tags on-colony (BRNW 2012). One explanation for this discrepancy could be that terns returning from the Columbia River forage in other water bodies along the way and return to the colony with non-salmonid prey from these secondary foraging areas. There was no indication that Caspian terns were regularly foraging on the way to or from the Columbia or Snake Rivers. Only 7 out of 39 trips (18%) outside of the Potholes Reservoir area included foraging at a secondary destination; in all but one case this foraging occurred during the out-going portion of the trip. The commuting portions of foraging trips were relatively straight; on average the path taken was only 4 % longer than a straight line, indicating that terns travel directly between primary foraging areas and the colony without changing course to search for prey at other water bodies along the way. The core commuting routes between Potholes Reservoir and the Columbia River passed over potential foraging sites (in particular the Frenchman Hills Wasteway), but did not follow waterways that connect the reservoir and the river, like Lower Crab Creek.

Terns foraging at the rivers usually made one trip per day, while terns foraging around Potholes Reservoir made two or three trips per day. Even if the same numbers of terns are foraging in each area and returns with a fish bill load from every trip, there will be 2 – 3 more bill loads from the Potholes Reservoir area than from the rivers. Trips to the rivers also included more foraging time and more foraging bouts than trips within the Potholes area; therefore, terns likely consumed more fish per trip to the Columbia or Snake river, but each trip can only result in one bill load fish brought back to the colony. These differences in foraging behavior between those terns that foraged on the Columbia or Snake river and those that foraged in the Potholes

Reservoir area could explain why predation rates on juvenile salmonids based on bill load observations have been consistently lower than those based on smolt PIT tag recoveries on-colony. Differences due to foraging dynamics could be exaggerated if terns foraging on salmonids in the Columbia or Snake river are less likely to return with a bill load because it requires more energy to carry a bill load that much further, or because salmonids are more likely to be kleptoparasitized by gulls at the colony.

In summary, our study found that just over half of the terns tagged at the Goose Island colony were foraging in the Columbia River, and that terns showed consistency in their foraging destinations. Trip characteristics and the number of trips per day were different depending on foraging destination. We also found that a small proportion of the tagged terns with active nests were foraging at the Snake River; previously, it had been assumed that PIT tags from Snake River salmonids recovered on Goose Island came from non-breeding terns. These trips to the Snake River are the longest ever recorded for a breeding Caspian tern. Foraging activity along the Columbia River was concentrated in the Wanapum and Priest Rapids dam reservoirs and around islands in the Hanford reach. Most of the foraging areas identified on the rivers and close to Potholes Reservoir were also associated with core resting areas. Finally, we did not find any evidence that terns regularly foraged in other water bodies on the way to or from the Columbia and Snake rivers. Commuting routes to the Columbia River were generally direct. However, the difference in the frequency of trips to the Columbia River versus trips within the Potholes Reservoir area could explain why bill load observations underestimate tern predation rates on salmonids, relative to recoveries of smolt PIT tags at this colony. While our results have provided new insight into the foraging behavior of Caspian terns breeding at the Goose Island colony, it is important to acknowledge that these data were collected over a relatively short time period. We would expect foraging behavior to change throughout the breeding season in response to changing energy requirements and prey availability, and to change between breeding seasons in response to different prey conditions and colony size.

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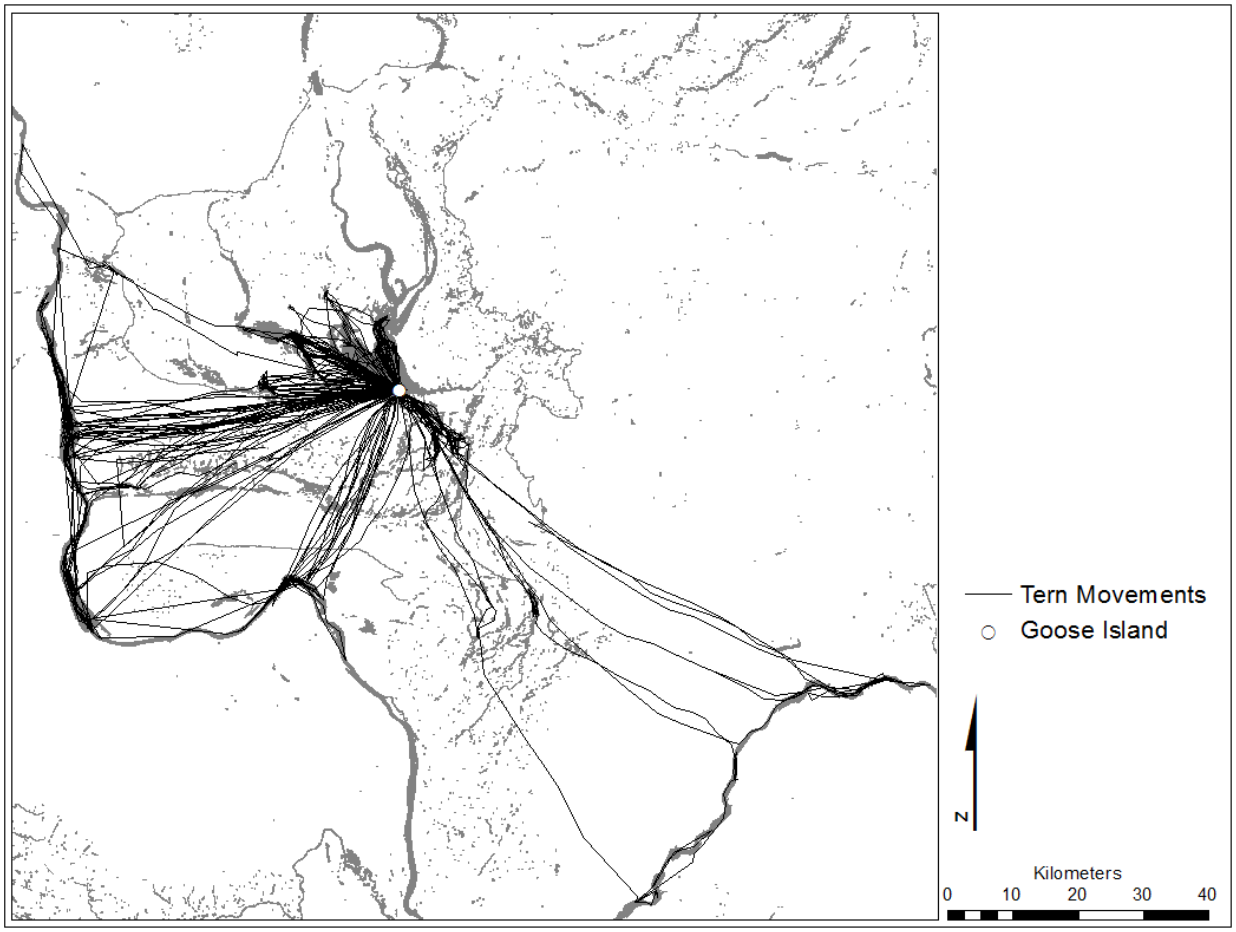


Figure C1. All movements recorded from Caspian terns equipped with GPS data-loggers and nesting at Goose Island in Potholes Reservoir, Washington, during the 2013 nesting season.

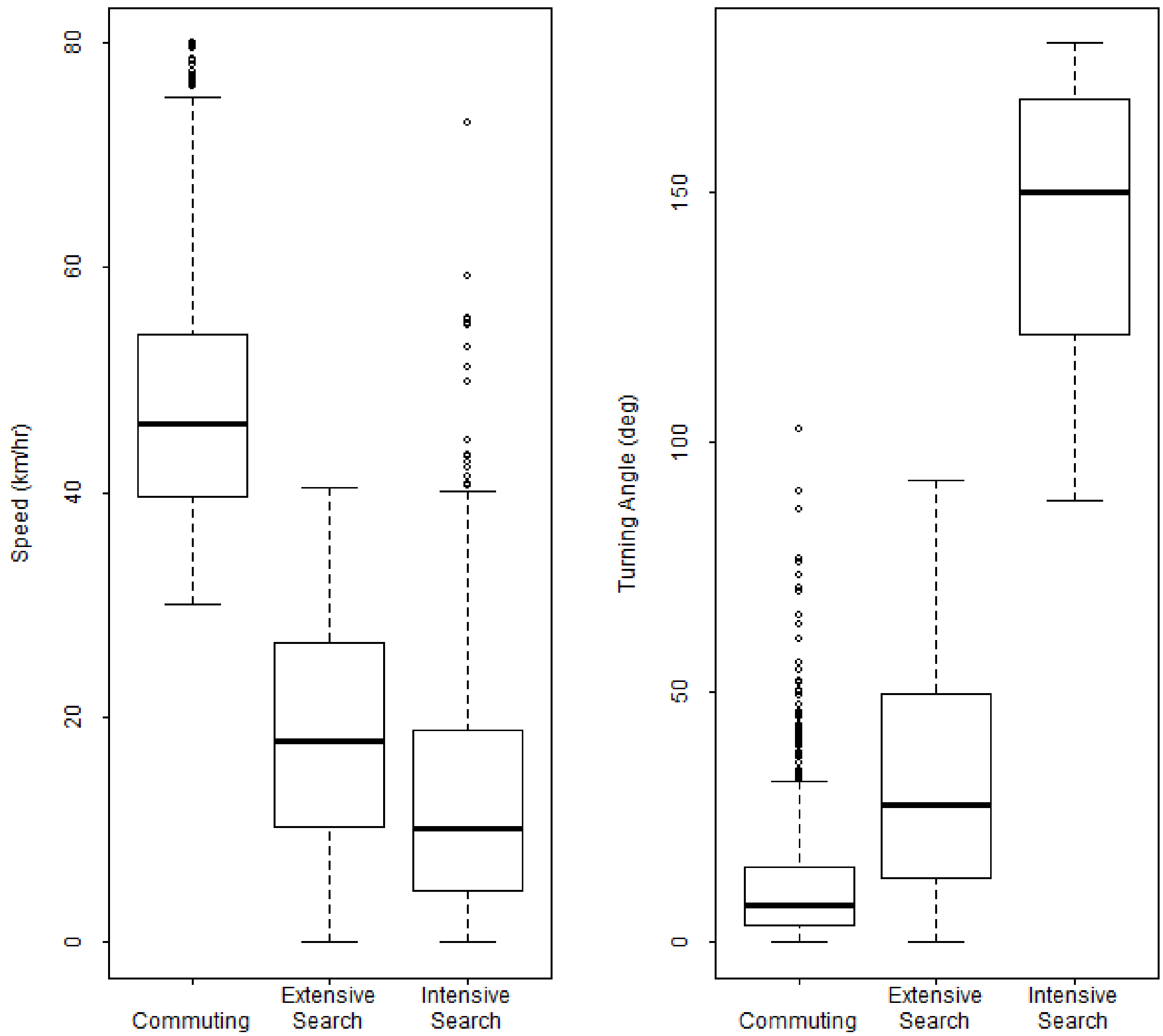


Figure C2. Plots of velocity and turning angle for the three movement states as defined by cluster analysis. Data on movement characteristics are for Caspian terns equipped with GPS data-loggers and nesting at Goose Island in Potholes Reservoir, Washington, during the 2013 nesting season.

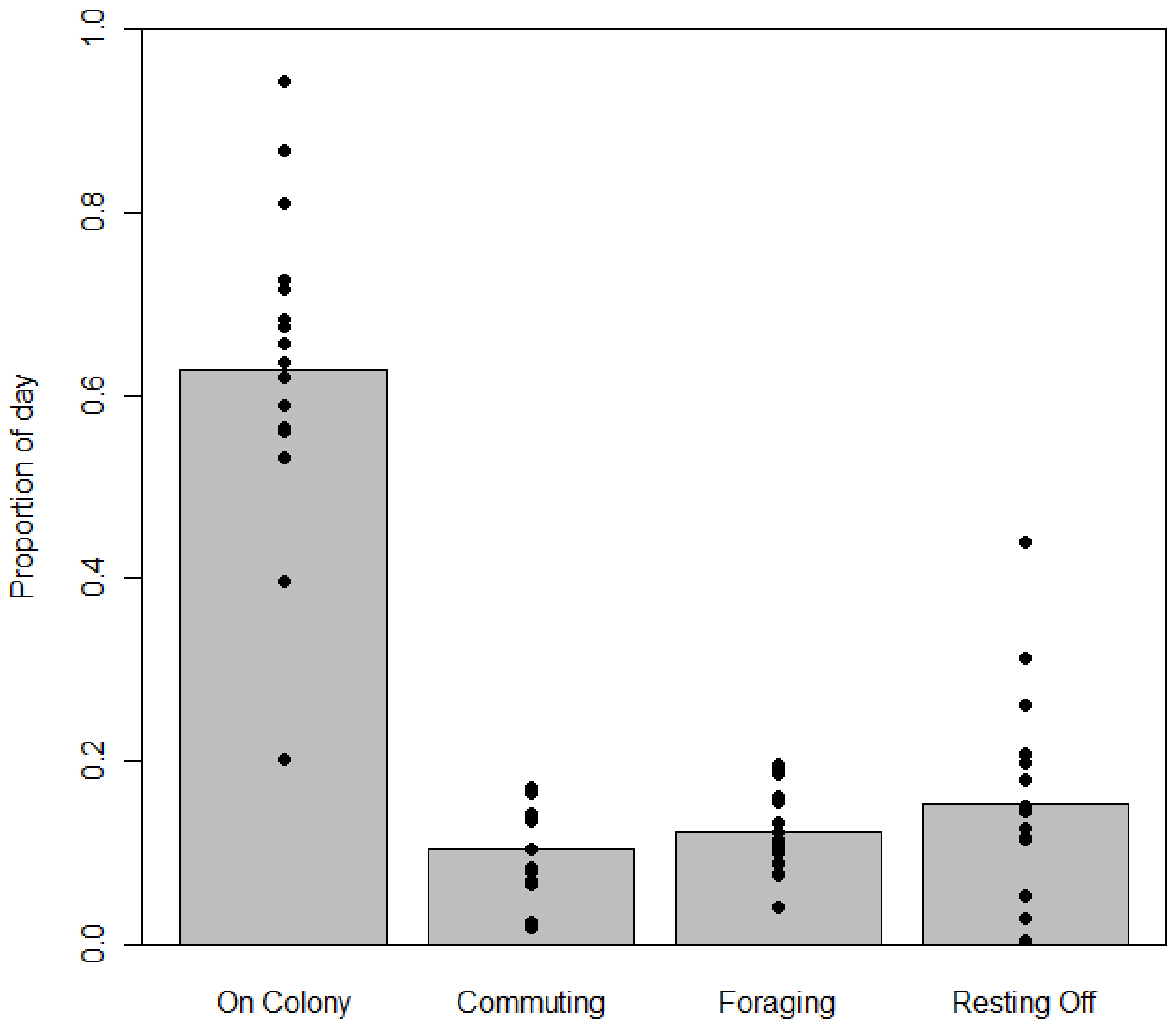
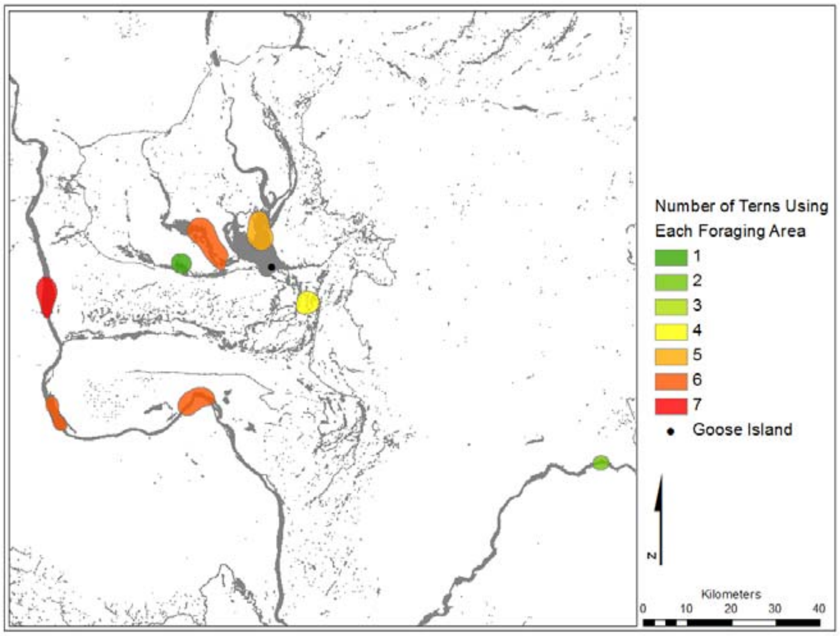


Figure C3. Proportion of the day spent on-colony, commuting, foraging, and resting off-colony for 18 Caspian terns tracked with GPS data-loggers while attending nests with eggs at the Goose Island colony on Potholes Reservoir, Washington, during the 2013 nesting season. Bar heights are means for all terns combined and black dots are averages for each individual.



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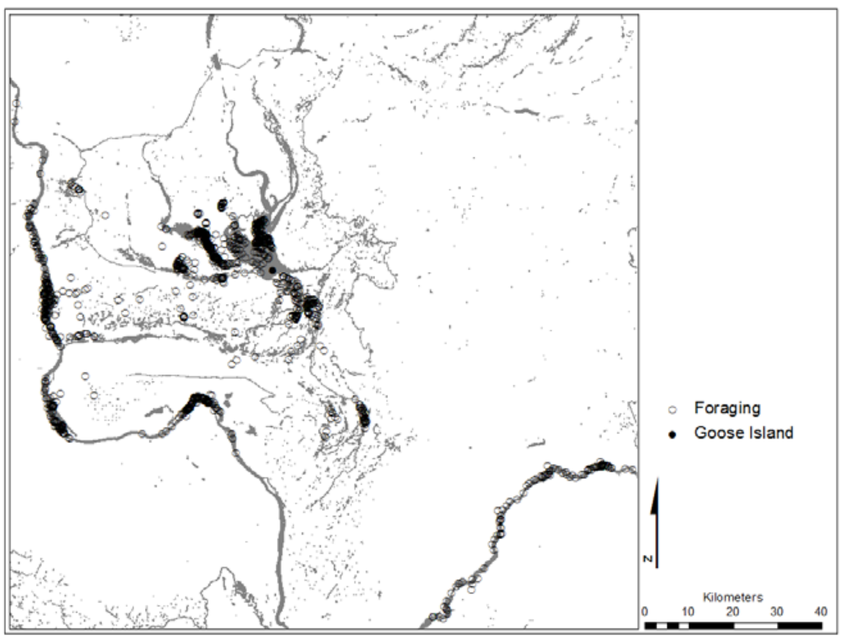
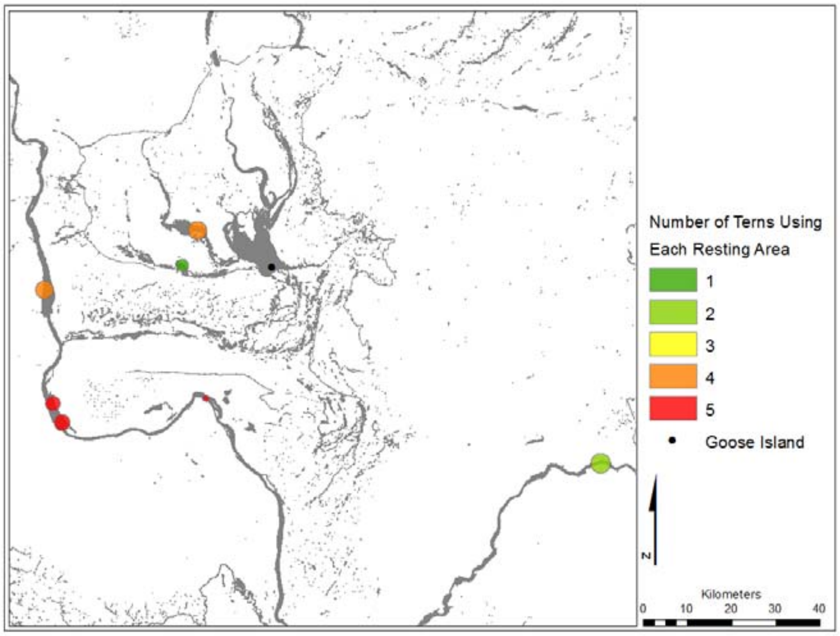


b

Figure C4. Maps of the core foraging areas used by Caspian terns breeding at Goose Island, Potholes Reservoir, WA, during the 2013 nesting season. The top map (a) shows the 50 percent volume polygons for kernel density estimate (kde) based on all foraging locations more than 2.5 km from the colony; colors indicate the number of individual terns that used each area. The bottom map (b) shows all the foraging locations used to develop estimates.



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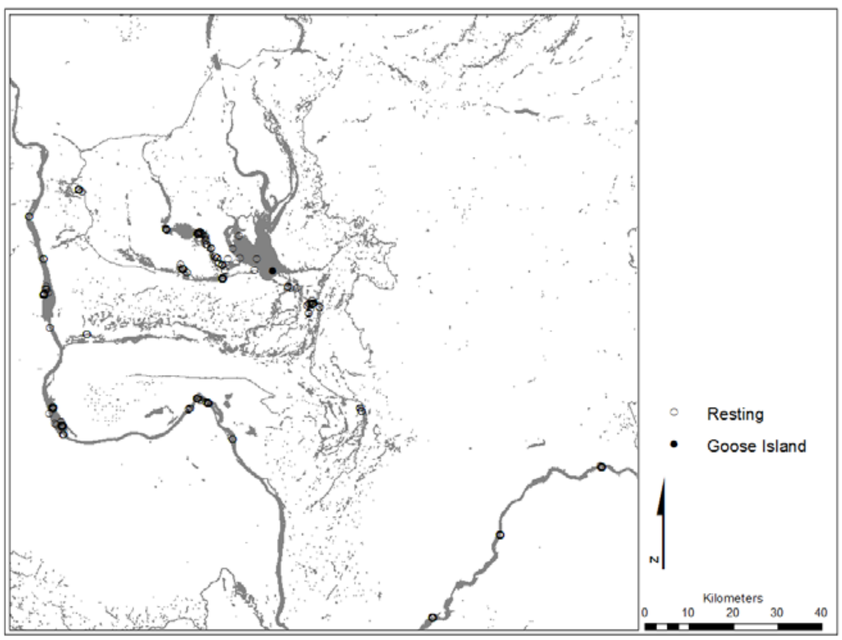

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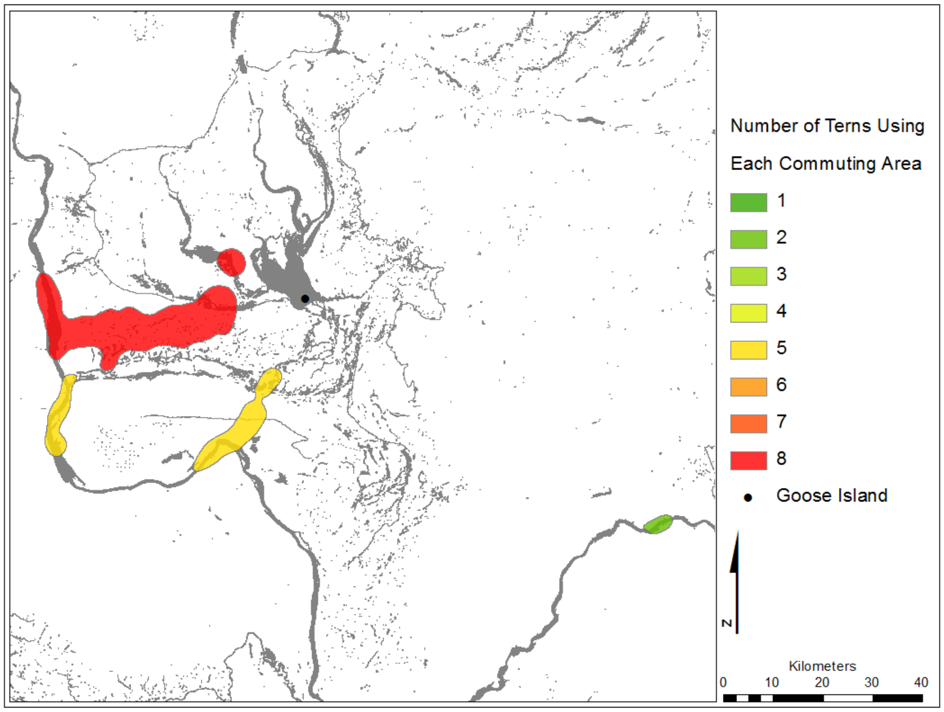
Figure C5. Maps of the core resting areas used by Caspian terns breeding at Goose Island, Potholes Reservoir, WA, during the 2013 nesting season. The top map (a) shows the 50 percent volume polygons for kernel density estimate (kde) based on all resting locations more than 2.5 km from the colony; colors indicate the number of individual terns that used each area. The bottom map (b) shows all the resting locations used to develop estimates.



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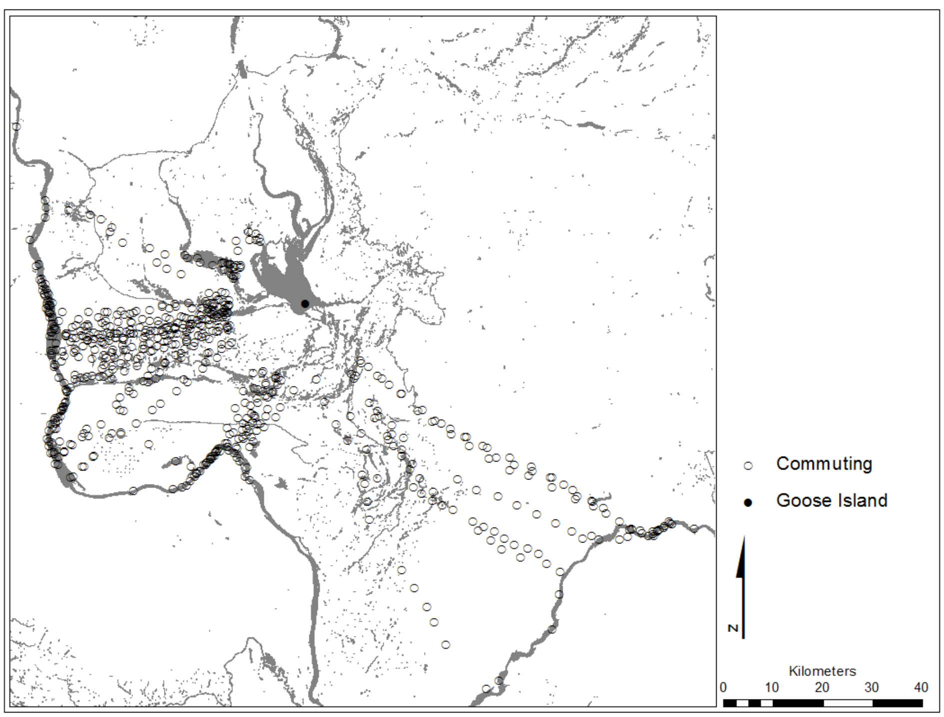

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Figure C6. Maps of the core commuting routes used by Caspian terns breeding at Goose Island, Potholes Reservoir, WA, during the 2013 nesting season. The top map (a) shows the 50 percent volume polygons for kernel density estimate (kde) based on all commuting locations more than 15 km from the colony; colors indicate the number of individual terns that used each area. The bottom map (b) shows all the commuting locations used to develop estimates.

Table C1. Summary of data collection from GPS data-logger tagged Caspian terns from the colony on Goose Island, Potholes Reservoir, WA, during the 2013 nesting season. Table shows the number of days and trips when data were collected and the number of terns that contributed to each data set. Data for a particular tern day were considered complete if more than 60% of attempted locations were successful. Trips were considered complete if all gaps of more than three locations (16 min) did not add up to more than 25% of the total trip duration. Statistical analyses were limited to complete days and trips by breeding adult terns that were attending nests with eggs.

	Number of Terns Tracked	Sample Size
Days		
Total Days	23	67
Complete Days	20	53
Complete Days for Nests with Eggs	18	49
Trips		
Total Trips	22	97
Complete Trips	22	83
Complete Trips for Nests with Eggs	19	75

Table C2. Table showing all trips by GPS data-logger tagged Caspian terns from the colony on Goose Island, Potholes Reservoir, WA, during the 2013 nesting season. Individuals are categorized by sex and nesting status. Trips are grouped by primary destination. The table includes all complete and partial trips.

			Number of Trips by Destination				
Sex	Bird ID	Nest Status	Columbia River	Potholes	Intermediate	Snake River	Total Trips
Females							
	CT04	Eggs	5				5
	CT07	Eggs	5				5
	CT09	Eggs	4				4
	CT14	Eggs				2	2
	CT19	Eggs		1	1		2
	CT21	Eggs	4	1			5
	CT24	Chicks		6			6
	CT27	Eggs	2				2
	SK1	Eggs	1	1			2
Males							
	CT01	Eggs	3				3
	CT02	Eggs	2				2
	CT08	Eggs	3	6			9
	CT12	Eggs		1			1
	CT13	Eggs	1	8			9
	CT15	Eggs				1	1
	CT20	Eggs	2				2
	CT22	Chicks		2			2
	CT25	Eggs		7	1		8
	CT26	Eggs		8	1		9
	SK2	Eggs/ Chicks		6			6
	SK3	Failed				1	1
	SK4	Eggs		11			11
Total Trips			32	58	3	4	97

Table C3. Summary table showing the number of trips taken per day by trip destination for Caspian terns nesting on Goose Island, Potholes Reservoir, WA, during the 2013 nesting season. This table is based on 53 complete tracking days for a total of 20 Caspian terns. There were 4 days when a tern was recorded not making any trips from the colony; those data are not included in this table.

	Number of trips per day		
	1	2	3
Columbia River	16	3	0
Potholes Reservoir	2	8	9
Snake River	4	0	0
Multiple Sites	0	5	2

Table C4. Characteristics of foraging trips made by Caspian terns tracked with GPS data-loggers while attending nests with eggs at the colony on Goose Island, Potholes Reservoir, WA, during the 2013 nesting season. Trip characteristics are summarized by primary trip destination. Values reported are observed means, with the range in parentheses. Differences in trip characteristics were analyzed using mixed-effects models and generalized linear mixed models (foraging patches) that included a random effect of individual. F-tests are reported for mixed-effects models and χ^2 test is reported for the generalized linear missed model.

Characteristic	Columbia River (n = 24)	Potholes (n = 52)	Intermediate (n = 3)	Snake River (n = 4)	Test Statistic	p-value
Trip Duration (min)	355 (123 – 772)	115 (28 – 264)	144 (48 – 280)	726 (551 – 940)	32.65	<0.001
Foraging Patches	2.4 (1 – 4)	1.5 (0 – 3)	2.0 (1 – 4)	4.0 (4 – 6)	21.38	<0.001
Foraging (min)	121 (24 – 245)	54 (0 – 121)	91 (28 – 145)	175 (88 – 324)	12.72	<0.001
Commuting (min)	96 (20 – 204)	22 (4 – 48)	46 (20 – 75)	153 (112 – 184)	45.95	<0.001
Resting (min)	140 (0 – 347)	38 (0 – 152)	20 (0 – 60)	396 (247 – 684)	17.09	<0.001

Table C5. Correlations between average daily activity rates for 18 Caspian terns tracked with GPS data-loggers while attending nests with eggs at the colony on Goose Island, Potholes Reservoir, WA, during the 2013 nesting season. P-values for Spearman's correlation tests are above the diagonal and estimates of the correlation coefficient (ρ) are below the diagonal; significant relationships are shown in italics.

Behavior	On-Colony	Foraging	Commuting	Resting Off-Colony
On-Colony	--	0.16	*< 0.001*	*< 0.001*
Foraging	-0.35	--	0.59	0.66
Commuting	*-0.77*	0.14	--	*0.006*
Resting Off-Colony	*-0.89*	0.11	*0.62*	--

APPENDIX D: Supplemental Figures and Tables

This appendix contains summary figures and tables and other ancillary information requested during the review of an earlier draft of this report.

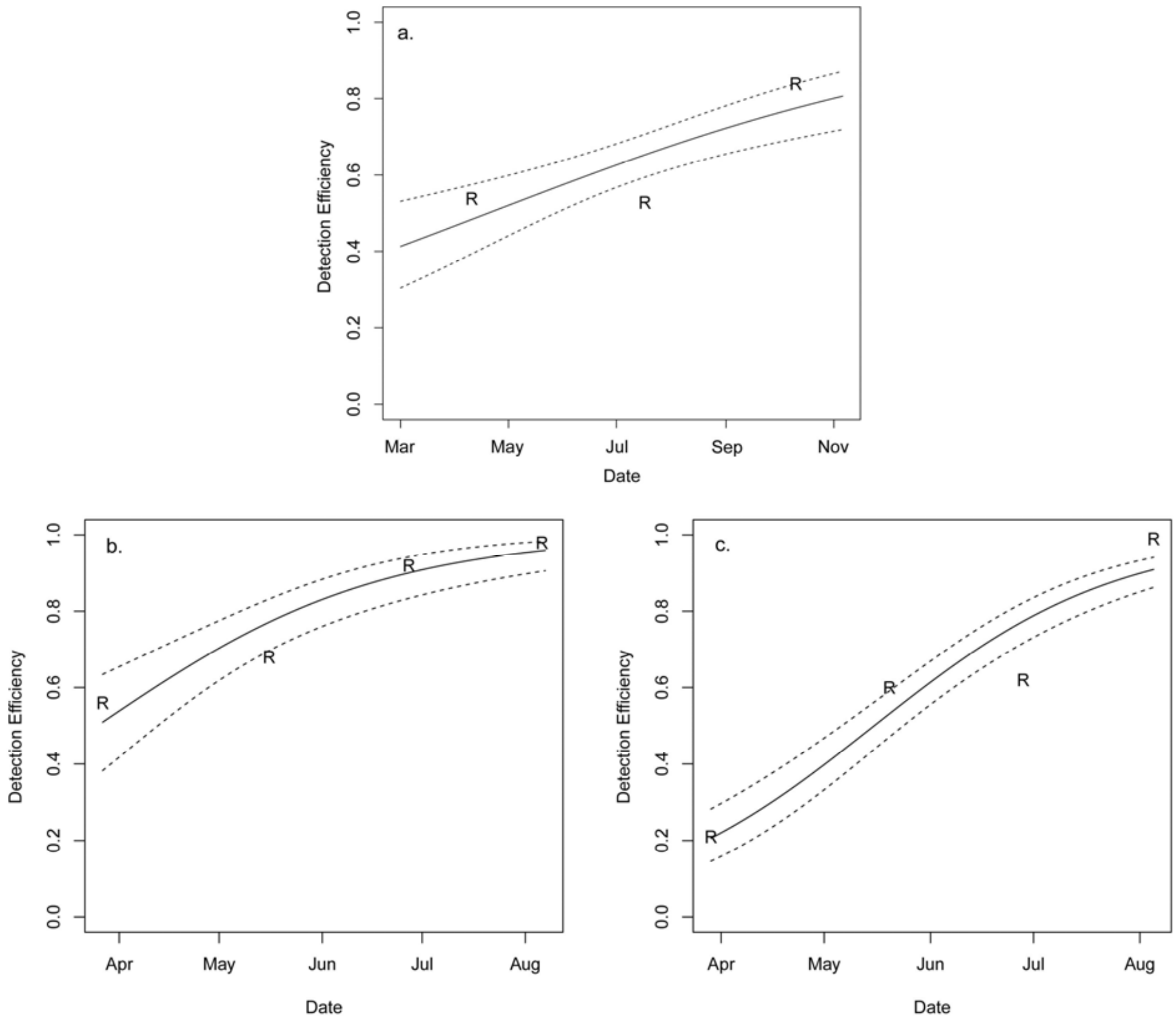

Figure D1. PIT tag detection efficiency estimates at the East Sand Island (a), Crescent Island (b) and Goose Island (c) Caspian tern colonies in 2013. Solid and dashed lines represent the best fit and 95% confidence bounds from logistic regression. Actual proportion of control tags recovered denoted by 'R' (see also Table 3).

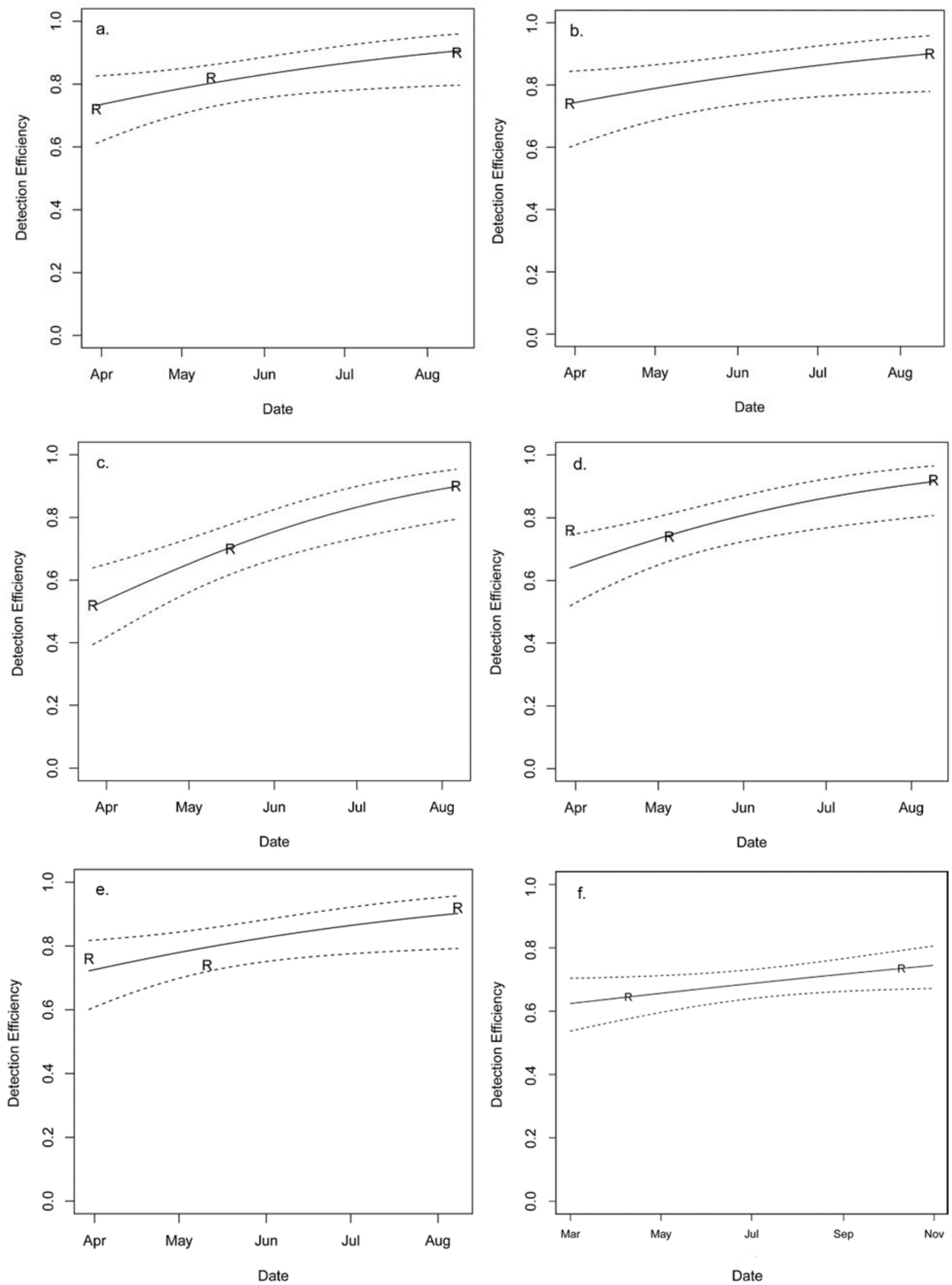

Figure D2. PIT tag detection efficiency estimates at the Miller Rocks Island California and ring-billed gull colony {a is for standard PIT tags and b is for PIT tags encapsulated in polyolefin tubing}, Crescent Island (c), Island 20 (d), Blalock Island (e), and the East Sand Island double-crested cormorant colony (f) in 2013. Solid and dashed lines represent the best fit and 95% confidence bounds from logistic regression. Actual proportion of control tags recovered denoted by 'R' (see also Table 3).

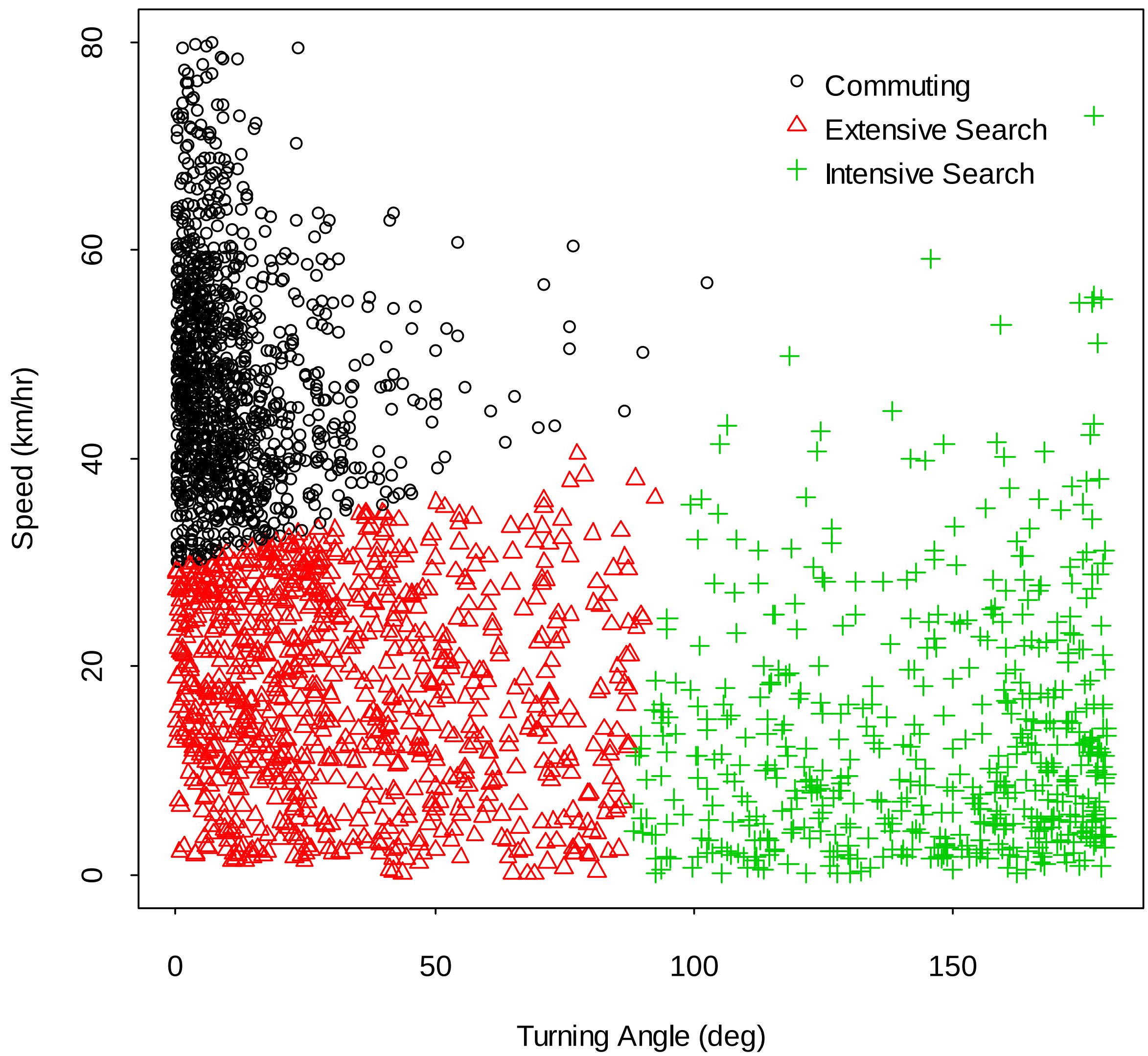

Figure D3. Plot showing the groupings identified by k-means cluster analysis as a function of speed (km/hr) and turning angle (deg) of each active GPS location.

Table D1. Estimated colony size (number of breeding pairs) and nesting density (nests/m^2) for Caspian Terns nesting on East Sand Island in the Columbia River estuary during 2000-2013. Errors in the estimates are 95% confidence intervals (CI).

Year	Colony Size	Lower 95% CI	Upper 95% CI	Nesting Density	Lower 95% CI	Upper 95% CI
2000	8,513	7,597	9,429	0.62	0.55	0.69
2001	8,982	8,427	9,537	0.57	0.53	0.61
2002	9,933	9,552	10,314	0.55	0.53	0.57
2003	8,325	7,838	8,812	0.45	0.42	0.48
2004	9,502	8,905	10,099	0.50	0.47	0.53
2005	8,822	8,325	9,319	0.45	0.42	0.48
2006	8,929	8,188	9,670	0.55	0.50	0.60
2007	9,623	8,880	10,366	0.70	0.65	0.75
2008	10,668	9,923	11,413	0.72	0.67	0.77
2009	9,854	9,509	10,199	0.70	0.68	0.72
2010	8,283	7,412	9,154	0.70	0.63	0.77
2011	6,969	5,759	8,179	0.85	0.75	0.95
2012	6,416	5,545	7,287	1.06	0.92	1.20
2013	7,387	6,776	7,998	1.17	1.06	1.28

Table D2. Estimated colony size (number of breeding pairs) for Caspian terns nesting at the nine Corps-constructed tern islands in interior Oregon and northeastern California in 2008-2013. A blank indicates that the island was not available for tern nesting (i.e., either not yet constructed or constructed and subsequently dismantled) during that year. A zero indicates that the island was available but not used by nesting terns during that year.

Year	Fern Ridge[1]	Crump Lake[2]	East Link[3]	Dutchy Lake[3]	Gold Dike[3]	Tule Lake[4]	Orems Unit[5]	Sheepy Lake[5]	Malheur Lake[6]	**TOTAL**
2008	0	428								428
2009	0	697	7	8						712
2010	0	71	29	0	0	0	0	258		358
2011	0	35	2	0	0	34	2	188		261
2012	0	115	10	0	4	207	0	212	232	780
2013	0	223	21		0	79	0	316	530	1,169

[1] In Fern Ridge Reservoir near Eugene, Oregon
[2] In the Warner Valley near Adel, Oregon
[3] In the Summer Lake Wildlife Area near Summer Lake, Oregon
[4] In Tule Lake National Wildlife Refuge near Tuklelake, California
[5] In Lower Klamath National Wildlife Refuge near Klamath Falls, Oregon

Table D3. Anticipated pre- and in-season management strategies needed to facilitate successful nesting at the Caspian tern nesting islands that were built by the U.S. Army Corps of Engineers prior to the 2013 nesting season as part of the federal agencies' Caspian Tern Management Plan for the Columbia River Estuary (USFWS 2005, 2006). "**X**" denotes regular (yearly) need for strategy, while "x" denotes periodic need for strategy.

	Sheepy Lake	Tule Lake	East Link	Gold Dike	Malheur Lake	Crump Lake
	Pre-season Management Strategy					
Trap & remove mammals from island		**X**				
Dissuade gulls using human disturbance and/or bird netting on island	**X**		**X**			**X**
Vegetation removal on island	x	x	**X**	**X**	x	x
	In-season Management Strategy					
Lethal removal of gulls from island	**X**	**X**	**X**	**X**	**X**	**X**
Lethal removal of mammals from island	x	**X**	**X**	x	x	x
Locate and monitor owl nests near island	**X**	**X**	**X**	**X**	**X**	
Nocturnal colony monitoring	x	**X**	**X**	**X**	x	x
Trap & relocate owls visiting island	x	**X**	x	x	x	
Provide chick shelters on island	**X**	**X**	**X**	**X**	**X**	**X**

Table D4. Estimated proportional stock composition and 95% confidence intervals (CI) for Chinook salmon consumed by Caspian terns at East Sand Island in the Columbia River estuary in 2011- 2013. Data presented for the early (April/May; N = 57) and late (June/July; N = 42) outmigration periods. Confidence intervals are derived from 100 bootstrap resampling's of baseline and mixture genotypes.

	Early(April/May)			Late (June/July)		
	Estimate	Lower 95% CI	Upper 95% CI	Estimate	Lower 95% CI	Upper 95% CI
Deschutes – Fall	0	0.0000	0.0000	0	0.0000	0.1268
West Cascades Tributary - Fall	0	0.0000	0.0340	0.1852	0.0383	0.3106
West Cascades Tributary - Spring	0.1023	0.0176	0.2062	0.0228	0.0000	0.0980
MCR & UCR - Spring	0.3599	0.2364	0.5378	0.0212	0.0000	0.0697
Spring Creek Group - Fall	0.1207	0.0517	0.1897	0.3257	0.1877	0.4754
Snake - Fall	0.0193	0.0000	0.0712	0.1777	0.0248	0.2943
Snake - Spring	0.2975	0.1293	0.4303	0.0233	0.0000	0.0698
UCR - Summer/Fall	0.0496	0.0000	0.1033	0.1784	0.0504	0.3117
Willamette - Spring	0.0507	0.0000	0.0690	0	0.0000	0.0210
Rogue	0	0.0000	0.0000	0.0657	0.0000	0.1145

Table D5. Estimated proportional stock composition and 95% confidence intervals (CI) for steelhead consumed by Caspian terns (N =100) and double-crested cormorants (N = 63) at East Sand Island in the Columbia River estuary in 2011- 2013. Confidence intervals are derived from 100 bootstrap resampling's of baseline and mixture genotypes.

	Caspian terns			Double-crested cormorants		
	Estimate	Lower 95% CI	Upper 95% CI	Estimate	Lower 95% CI	Upper 95% CI
Lower Columbia	0.2168	0.1286	0.3034	0.206	0.0915	0.3177
Mid Columbia	0.1249	0.0572	0.2575	0.1619	0.0785	0.3086
Snake River	0.5621	0.4351	0.6381	0.4849	0.2971	0.5718
Upper Columbia River	0.0233	0.0000	0.0835	0.0818	0.0013	0.1755
Willamette River	0.0679	0.0221	0.1136	0.0654	0.0000	0.1388
Washington Coast	0.0052	0.0000	0.0296	0	0.0000	0.0218

Table D6. Estimated proportional stock composition and 95% confidence intervals (CI) for coho salmon consumed by Caspian terns (N =20) and double-crested cormorants (N = 108) at East Sand Island in the Columbia River estuary in 2011- 2013. Confidence intervals are derived from 100 bootstrap resampling's of baseline and mixture genotypes.

	Caspian terns			Double-crested cormorants		
	Estimate	Lower 95% CI	Upper 95% CI	Estimate	Lower 95% CI	Upper 95% CI
Washington Coast	0.0878	0.0000	0.3165	0.0139	0.0000	0.1187
Columbia River	0.8406	0.5651	0.9306	0.7435	0.5750	0.8021
North Oregon Coast	0.0716	0.0000	0.2683	0.2425	0.1434	0.3580

Table D7. Inter-colony movement rates of Caspian terns between 2012 and 2013. Data used in movement rate estimates were from terns banded as adults during 2005-2012 and re-sighted during 2006-2013. The numbers of individuals that moved in 2013 were estimated from movement rates multiplied by estimated numbers of adults present at source colonies in 2012.

Source colony	Receiving colony	Movement rates (%)	Estimated number of individuals that moved
East Sand Island	Crescent Island	0.4	56
East Sand Island	Goose Island	0.4	57
East Sand Island	Alternative sites	5.3	684
Crescent Island	East Sand Island	1.7	15
Crescent Island	Goose Island	2.8	23
Crescent Island	Alternative sites	5.1	43
Goose Island	East Sand Island	5.3	49
Goose Island	Crescent Island	9.9	92
Goose Island	Alternative sites	4.6	42
Alternative sites	East Sand Island	1.6	25
Alternative sites	Crescent Island	1.6	25
Alternative sites	Goose Island	5.2	81

Table D8. Estimated colony size (number of breeding pairs) and nesting density (nests/m^2) for double-crested cormorants nesting on East Sand Island in the Columbia River estuary during 1997-2013. Errors in the estimates are 95% confidence intervals (CI). Dashes indicate that data are not available.

Year	Colony Size	Lower 95% CI	Upper 95% CI	Nesting Density	Lower 95% CI	Upper 95% CI
1997	5,023	4,722	5,324	0.98	--	--
1998	6,285	5,908	6,662	1.08	--	--
1999	6,561	6167	6,955	1.03	--	--
2000	7,162	6,732	7592	1.02	--	--
2001	8,120	7,633	8,607	1.14	--	--
2002	10,230	9616	10844	1.09	--	--
2003	10,646	10,007	11,285	0.89	--	--
2004	12,480	11,731	13,229	0.73	--	--
2005	12,287	11,550	13,024	1.20	--	--
2006	13,738	12,914	14,562	1.10	--	--
2007	13,771	12,945	14,597	1.23	--	--
2008	10,950	10,585	11,315	1.07	--	--
2009	12,087	11,929	12,245	1.23	--	--
2010	13,596	13,130	14,062	1.28	--	--
2011	13,045	12,881	13,209	1.15	--	--
2012	12,301	11,886	12,716	1.23	--	--
2013	14,916	14,545	15,287	1.20	--	--

Table D9. Estimated proportional stock composition and 95% confidence intervals (CI) for Chinook salmon consumed by double-crested cormorants at East Sand Island in the Columbia River estuary in 2011- 2013. Data presented for the early (April/May; N = 34) and late (June/July; N = 31) outmigration periods. Confidence intervals are derived from 100 bootstrap resampling's of baseline and mixture genotypes.

	Early(April/May)			Late (June/July)		
	Estimate	Lower 95% CI	Upper 95% CI	Estimate	Lower 95% CI	Upper 95% CI
Deschutes – Fall	0	0.0000	0.0279	0.0318	0.0000	0.0967
West Cascades Tributary - Fall	0	0.0000	0.0904	0.3234	0.1937	0.6022
West Cascades Tributary - Spring	0	0.0000	0.0937	0	0.0000	0.1072
MCR & UCR - Spring	0.2148	0.0700	0.4111	0	0.0000	0.0645
Spring Creek Group - Fall	0.1176	0.0243	0.2059	0.4827	0.1895	0.5430
Snake - Fall	0.036	0.0000	0.1603	0	0.0000	0.0651
Snake - Spring	0.4636	0.1888	0.6229	0.0323	0.0000	0.0968
UCR - Summer/Fall	0.168	0.0181	0.2859	0.0653	0.0000	0.2257
Willamette - Spring	0	0.0000	0.0000	0	0.0000	0.0000
Rogue	0	0.0000	0.0000	0.0645	0.0000	0.1322